The London School of Economics and Political Science

CHOOSING ENVIRONMENTAL POLICY INSTRUMENTS:

CASE STUDIES OF MUNICIPAL WASTE POLICY IN SWEDEN AND ENGLAND

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I certify that the thesis I have presented for examination for the PhD degree of the London School of Economics and Political Science is solely my own work other than where I have clearly indicated that it is the work of others (in which case the extent of any work carried out jointly by me and any other person is clearly identified in it).

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Abstract

European governments have during the last couple of decades shown an interest in new types of environmental policy instruments (EPIs) such as environmental taxes, tradable permit schemes and voluntary approaches, as opposed to relying on traditional forms of regulation. The interest in so-called ‘new’ EPIs (NEPIs) has led many governments to commit both to a more diverse EPI mix and to a policy process characterised by procedural rationality, in terms of considering a wide range of alternative instruments and assessing them in a systematic and transparent way.

The first aim of this thesis is to examine the success of the quest for NEPIs at the national level in the field of municipal waste policy in two countries; the UK (England) and Sweden. In addition to mapping out EPI diversity, two contrasting theories on the pattern of adoption of instruments over time are evaluated, specifically focusing on the degree of coercion associated with EPIs. It is found that the waste policy mix in England has become more diverse, while the Swedish mix is characterised by a higher degree of coercion.

The second aim is to analyse whether the instrument choice process has become more procedurally rational, and, if so, conducive to the adoption of NEPIs. A range of instrument choice theories at the macro-, meso- and micro-levels drawn from the public policy and political science literature are used to explain whether the ideal of procedural rationality is achievable or not. A case study methodology is used, in which the processes leading to the landfill allowance trading scheme (LATS) in England and and the waste incineration tax in Sweden are studied. It is found that the procedural rationality was higher in the England case, but that it is not a necessary nor sufficient cause for adoption of a NEPI.

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Acronyms and abbreviations

BMW	Biodegradable municipal waste
BPEO	Best practicable environmental option
BRAS	*Swedish abbreviated name for the 2003 inquiry on a waste incineration tax, i.e. the 2003 BRAS inquiry*
CBA	Cost-benefit analysis
CBI	Confederation of British Industries
CCA	Compliance Cost Assessment
CHP	Combined heat and power
CIWM	Chartered Institute for Waste Management
CO2	Carbon dioxide
CSE	Confederation of Swedish Enterprise (*Sw. Svenskt Näringsliv*)
DEFRA	Department for Environment, Food, and Rural Affairs
DETR	Department for Environment, Transport, and the Regions
DoE	Department of Environment
DTI	Department of Trade and Industry
EA	Environment Agency for England and Wales
EAP	Environmental Action Programme
EC	European Community
EEA	European Environment Agency
EEC	European Economic Community
EfW	Energy from Waste
EfWA	Energy from Waste Association
EPI	Environmental policy instrument
ESA	Environmental Services Association
ETS	Emissions Trading System
EU	European Union
EU-15	The fifteen EU member states before the 2004 enlargement
EUR	Euro
GBP	Great Britain Pounds
GDP	Gross domestic product
GHG	Greenhouse gases
HoC	House of Commons
HoL	House of Lords
IPP	Integrated product policy
IPPC	Integrated pollution prevention and control
LATS	Landfill Allowance Trading Scheme
LAWDC	Local authority waste disposal company
LCA	Life-cycle analysis
LGA	Local Government Association
LTCS	Landfill Tax Credit Scheme
MoE	Ministry of Environment
MoF	Ministry of Finance
MoI	Ministry of Industry, Employment and Communications
MP	Member of Parliament
MW	Municipal waste
NIC	National insurance contribution

NEPI	New environmental policy instrument
NGO	Non-governmental organization
NOx	Nitrogen oxides
ODPM	Office of the Deputy Prime Minister
OECD	Organisation for Economic Co-operation and Development
PCBs	Polychlorinated biphenyl
PET	*Swe. Polyetylentereftalat* (recycled plastic used for bottles)
PPG	Planning Policy Guidance
R&D	Research & Development
RDF	Refuse-Derived Fuel
RIA	Regulatory Impact Assessment
RVF	Svenska Renhållningsverksföreningen (*Eng. The Swedish Association of Waste Management*)
SALA	Swedish Association of Local Authorities (*Sw. Sveriges Kommuner och Landsting*)
SEK	Swedish *Krona* (1 EUR ~ 9 SEK)
SEPA	Swedish Environmental Protection Agency (*Sw. Naturvårdsverket)*
SFS	Svensk Författningssamling (*Eng. The Swedish Code of Statutes*)
SKL	Sveriges Kommuner och Landsting (*Eng. Swedish Association of Local Authorities and Regions*)
SN	Svenskt Näringsliv (*Eng. Confederation of Swedish Enterprise*)
SNF	Svenska Naturskyddsföreningen (*Eng. Swedish Society for Nature Conservation*)
SRIA	Swedish Recycling Industries´Association (*Sw. Återvinningsindustrierna*)
SSNC	Swedish Society for Nature Conservation (*Sw. Svenska Naturskyddsföreningen*)
SWM	Swedish Waste Management (*Sw. Avfall Sverige, previously Renhållningsverksföreningen RVF*)
TPS	Tradable permit system
UK	United Kingdom
US	United States
VA	Voluntary agreement
VAT	Value-added tax
WIP	Waste Implementation Programme
WRAP	Waste and Resources Action Programme

“Western culture is a culture of hope. We can easily live in two worlds simultaneously: the world as we believe it to be, and the world as we think it ought to be. These worlds need not have much in common. But the fact that ‘is’ and ‘ought to’ often diverge does not cause us to abandon faith in one or the other. We tend to behave in the practical world on its terms, without abandoning our ideals. Even when we notice discrepancies between the two worlds, we tend to reconcile them through hope: We hope to bring about agreement between the way things are and the way things ought to be.”

– Nils Brunsson (2006, p. 11) on the dream of rationality

“This is a creative period in environmental policy. Policy makers reassess their goals (e.g., from pollution control to pollution prevention, risk management, or sustainable development), apply novel policy instruments, consider how to integrate across environmental programs and policy sectors, and think more ecologically and globally.”

– Daniel J. Fiorino (1999, p. 466)

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Chapter 1

Introduction: The Quest for New Instruments

1.1 Towards a wider range of environmental policy instruments

As an instrument for meeting the challenging UK greenhouse gas emission targets, the then UK Environment Minister Elliot Morley admitted in July 2005 that the government could consider personal carbon quotas (Anon. 2005b). Having individuals swipe a carbon quota card every time they buy products such as petrol, utility bills and flights as a key policy instrument for mitigating climate change represents a far step from traditional environmental regulation, in the form of emission or quality standards targeting industry. The Minister argued that "[w]e should have an open mind about the kind of levers that we apply and not be afraid to think the unthinkable" (*ibid*.). The interest expressed in this new instrument idea can be seen as a recent example in the quest for new policy instruments to achieve environmental targets and objectives, that was started in the 1970s and intensified in the 1980s and 1990s. The purpose of this thesis is to take stock of the achievements in this quest and study the openness of policy-makers to new instruments when they formulate environmental policy.

European countries have during the last three decades shown considerable interest in trying out new types of environmental policy instruments (EPIs) and innovative instrument designs to achieve environmental objectives and targets, as opposed to relying on traditional forms of environmental regulation (see Golub 1998a; Jordan, Wurzel et al. 2003c; Tews, Busch et al. 2003). In parallel to reconsidering *what* environmental policy should achieve – in terms of which environmental problems to prioritise, what activities and actors to address, and what objectives and targets to set – policy-makers have reconsidered *how* environmental policy should be implemented. Should the government achieve an environmental objective by *prescribing* certain actions to be taken? Should it try to modify the *financial incentives* facing firms and individuals? Should it rely on providing *better information* for actors on which to base their decisions? These three alternative means to achieve an environmental objective are often seen as constituting a basic typology of public policy instruments; *regulatory instruments*, *financial incentives*, and *information transfer* (see e.g. Vedung 1998; de Bruijn and Hufen 1998). For environmental policy specifically, the OECD (2001c, p. 132) has defined a more detailed and empirically derived set of categories:

- Command-and-control instruments – e.g. licenses/permits, ambient quality standards, emission standards, process standards, prohibition bans
- Economic instruments – e.g. charges, taxes, tradeable permit systems, subsidies, deposit-refund systems
- Liability and damage compensation – e.g. strict liability rules, compensation funds, extended producer responsibility
- Education and information – e.g. public campaigns, technology diffusion, eco-labelling, publicity of sanctions for non-compliance
- Voluntary approaches – e.g. unilateral commitments, public voluntary schemes, negotiated agreements
- Management and planning – e.g. environmental management systems, zoning, land use planning

1.2 The depoliticisation of instrument choice?

Arguably, the choice of instrument among these six categories to address a given environmental problem has shifted from being perceived as highly ideologically charged to becoming increasingly seen as a technical design issue (see e.g. Gunningham and Gabrosky 1998; Huppes and Simonis 2000; Yachnin, Gagnon et al. 2000; Sterner 2003). This could be seen as a depoliticisation process, i.e. that instrument choice has moved from being a political or ideological concern to an administrative and professionalised problem-solving task. This contestable view is consistent with the rationalistic idea that policy objective and instrument (ends and means) can be separated, and that deciding on objectives (where and when to intervene in private behaviour, and to what extent) is more political and ideological than deciding on instruments (how to intervene) (see Howlett and Ramesh 1995). In other words, there is an expectation that

EPI choice could – and should – conform to the ideal of *procedural rationality*[1]. The choice of instrument from the widening range outlined above could thus be seen as a rather technical implementation decision, while decisions on policy objectives, overarching principles and the need for government intervention in the first place represent expressions of core ideological beliefs and political strategy (cf. Hall 1993). In other words, it could be argued that EPI choice has become one of finding the 'best-tool-for-the-job' (Böcher and Töller 2003).

There are several plausible reasons why such a shift in the nature of instrument choice might have occurred. First, the simple fact that new types of EPIs have appeared, both in theory and in practice, suggests that policy problem-solving has become more creative and innovative, and less ideologically, institutionally or politically constrained. Second, there is evidence to suggest that *a priori* commitments to and preferences for specific kinds of EPIs among actors have weakened (see e.g. Cook 2002; Golub 1998b). For example, environmental pressure groups were initially sceptical of economic instruments, as they were seen as unethically 'selling' the right to pollute. Industry, meanwhile, was supportive of the general idea of economic instruments and the flexibility they allow, but have proven to be less supportive when the financial burden they sometimes imply has been perceived as too large (Pearson 1995). The left-right affiliation of governments also seems less important, in that economic instruments and voluntary agreements have gained in popularity across the board, despite their origin in the neoliberal ideology (see e.g. Daugbjerg 2001a; Holm Pedersen 2003; Jordan, Wurzel et al. 2003a).

Third, the emphasis on a more rational and 'professional' policy-making process, including the choice of instruments to implement policy, has increased lately within the policy-maker community (see e.g. Cabinet Office 1999; Commission of the European Communities 2001b). This trend towards procedural rationality has been connected to the 'hollowing-out' of the state and increasing lack of trust in the political system, which supposedly leads to a stronger focus on procedures (as opposed to substantive outputs) within the state to rebuild trust (Howlett 2000b). In relation to the choice of

[1] Procedural rationality has been defined by Simon (1957 p. 76, in Parsons 1995 p. 278) as behaviour that is purposive or directed at realising goals of expressed values. Note that values here are considered as given, i.e. the values themselves are neither 'rational' nor 'irrational'. The concept of procedural rationality will be further discussed in chapter 3 of the thesis.

EPIs, an underlying assumption here is that an instrument choice process characterised by procedural rationality would be more conducive to the adoption of new forms of EPIs, or 'NEPIs', such as economic instruments, voluntary agreements, eco-labelling instruments (see Jordan et al. 2003b. p. 3). However, a procedurally rational choice process may not necessarily result in the *adoption* of a NEPI. This would depend on the specific choice criteria and the nature of the policy problem at hand. Instead, this thesis will examine the role of procedural rationality for the *consideration* of a wide range of alternative EPIs, whether resulting in a NEPI or an 'old' instrument type. An important formalisation of the procedural rationality ideal is Regulatory Impact Assessment (RIA). In such exercises, there is a strong element of first defining the problem, then formulating the objective and criteria, and, not until this is done, choose the most appropriate instrument from a set of identified alternatives.

However, this line of reasoning is in tension with those at the other end of the spectrum claiming that EPI choice is still highly politically controversial and loaded with value judgement, and effectively constrained by institutional biases. Again, there are several reasons why there might not have been a shift to more depoliticised and technical problem-solving. First, looking at the actual instrument choices made, it has been noted that traditional command-and-control regulation tends to persist, and there has not been a revolution of new EPIs – whether due to institutional or other constraints (Jordan, Wurzel et al. 2003a; Jordan, Wurzel et al. 2005c). Second, many argue that instrument choice is still a political game of winners and losers, due to the different cost/benefit distributions of alternative instruments. For example, Daugbjerg (1999, p. 163) states that "the choice of instruments in environmental policy is more contingent on political conditions than on technical considerations" (see also Macdonald 2001). Therefore, it would be unrealistic to assume a genuine interest in improving procedural rationality. Thirdly, numerous accounts of instrument choice conclude that the real-world choice process is far from the procedural rationality ideal (Majone 1976; Elmore 1987; Daugbjerg 1998b). For example, it has been argued that policy objectives are often vague, resulting in a 'means-end' rather than 'ends-means' process (Majone 1989). Furthermore, the actual range of potential EPIs considered may be significantly constrained. Linder and Peters (1989, p. 36) argue that

> "choices about the instruments of intervention are typically made without benefit of systematic, comparative assessments. Whether bound by the constraints of time, as in the case of the policy maker, or by disciplinary norms, as is the policy researcher, some instruments are simply

> deemed better than others by virtue of training, experience or precedent and are embraced accordingly".

Procedural rationality may thus be inherently impossible due to resource, cognitive or cultural barriers, or it may at best amount to *post-hoc* rationalisation of a decision. Böcher and Töller (2003, p. 3) go as far as calling the procedural rationality approach 'naïve instrumentalism'.

Whilst methodologically challenging, the question whether the quest for new EPIs in Europe has been associated with a depoliticisation of instrument choice needs to be investigated. In particular, it needs to be investigated within the municipal waste policy field, for reasons explained below (see section 1.5). In light of the two contrasting understandings of EPI choice outlined above, the aim of this thesis is to critically appraise the success of the quest for a more diverse EPI mix within municipal waste policy, including 'new' EPIs, in terms of choice outcomes. Furthermore, the success in translating the ideal of procedural rationality into practice in waste policy-making – whether it leads to the choice of a 'new' or an 'old' EPI – will be explored. The aim is thus to contribute to the EPI literature by analysing both instrument choice *outcomes* and *processes*. Two research questions guide this thesis:

- *Has the mix of EPIs for municipal waste policy become more diverse over time?*
- *What determines the instrument choice process? In particular, to what extent does the ideal of procedural rationality influence the consideration of alternative EPIs?*

These questions are examined empirically in the context of municipal waste policy in two European countries, Sweden and the United Kingdom (England), in the period 1995-2005. The first research question is addressed by a comparative compilation and review of the municipal waste policy instruments actually adopted over the last decade. The second research question is investigated through case studies of two recent instrument choice processes; the *landfill allowance trading scheme* introduced in England in 2005 and the *waste incineration tax* introduced in Sweden in 2006. Before presenting the aims and structure of the thesis further, the intentions stated by the governments in these two countries regarding the diversity of the EPI mix and the rationality of the instrument choice process need to be described.

1.3 Commitments to a diverse EPI mix and 'best-tool-for-the-job' approach

As stated above, there are two expectations arising from the quest for new EPIs. First, a wider range of different EPIs would be expected as an *outcome*. Second, a more comprehensive and rational ends-means policy *process* would be expected, to facilitate comprehensive consideration of EPIs and possible testing of new instruments. Regarding the former expectation, several environmental policy declarations at national and international levels have recognised the need for a diverse EPI mix. For example, the United Nations *Agenda 21* (UNCED 1992) addressed the need for extending the repertoire of EPIs.

> "What is needed is an appropriate effort to explore and make more effective and widespread use of economic and market-oriented approaches within a broad framework of development policies, law and regulation … [paragraph 8.30];
> In the near term, Governments should consider gradually building on experience with economic instruments and market mechanisms by undertaking to reorient their policies, keeping in mind national plans, priorities and objectives, in order to: (a) Establish effective combinations of economic, regulatory and voluntary (self-regulatory) approaches…[paragraph 8.32]"

At the European Union (EU) level, the *6th Environmental Action Programme* (EAP), adopted by the Parliament and Council in 2002, reinforced the commitment to continue the search for new instrument types and designs by stating that full consideration shall be given to "all available options and instruments" (Article 2.3)[2]. In particular, the Action Programme declares that instruments such as reforms of subsidies, tradable environmental permits, fiscal measures, standardisation activities, company environmental performance award schemes, and voluntary commitments or agreements shall be promoted (Articles 4 and 5).

At the national level – where the legal competence for adopting most policy instruments still lies – the last major environmental policy bill in Sweden exemplifies an awareness and commitment to using different types of EPIs[3].

> "In order to achieve the environmental quality objectives, effective instruments are required that enable the stimulation of a number of actions in society. In many cases attitudinal changes are required that lead to behavioural changes. The government is of the opinion that instruments need to be further developed to generate more effective action. Especially important are

[2] This commitment was also strongly conveyed in earlier EAPs. See Rittberger and Richardson (2003) for an EPI choice content analysis of the Fourth and Fifth EAPs respectively (adopted in 1987 and 1993).
[3] This environmental policy bill from 2005 was issued by the then Social Democratic government (Regeringens proposition 2005a). The new right-wing coalition government that entered into office in October 2006 has announced that it intends to increase the use of economic instruments and other new types of EPIs (Regeringskansliet 2006).

> economic instruments, like environmental taxes and charges and tradable emission rights, but also other market-based instruments, legal and administrative instruments, planning, and research, development and demonstration of new technology. This includes support to investments in and commercialisation of new technology that can contribute to the achievement of the environmental quality objectives. Targeted information can also be an effective instrument. Ecologically sustainable public purchasing is, in the view of the government, a powerful tool in this context that needs to be further developed" (Regeringens proposition 2005a, p. 229) (*author's translation*).

The Environment Agency (EA) in England and Wales has also demonstrated a strong awareness of diverse EPIs and recast its role in the sense of providing 'modern regulation'.

> "As times have changed, regulation has modernised too. Dialogue and joint problem solving, and the 'carrots' of incentive and reward are increasingly being used to supplement or replace the traditional 'stick' approach. Modern regulatory thinking has developed a wider and smarter range of tools than ever before – to include taxes, trading schemes, voluntary agreements and environmental management systems." (Environment Agency 2003, p. 1).

The key question, then, is whether these commitments to diversifying the EPI mixes have translated into practice. Previous studies have suggested that national repertoires have indeed diversified (Golub 1998a; OECD 2001c; EEA 2001), although this development is more of a 'steady evolution' than a 'sudden revolution' (Jordan, Wurzel et al. 2003a, p. 221). For example, it has been observed that in the case of environmental taxes, revenues represented on average almost 7% of total tax revenue in 21 OECD countries in 2001 (OECD 2003c). Daugbjerg and Svendsen (2001, p. 3) have estimated that the number of green taxes in OECD countries has increased from around 30 in 1987 to 110 in 1997. A more recent OECD publication on environmental taxes reported that there were now around 375 environmental taxes registered in its database, as well as around 250 charges and levies (OECD 2006, p. 26). In the case of voluntary agreements (VAs), surveys have shown that in the late 1990s there were more than 300 negotiated agreements in the EU-15 countries (EEA 1997, p. 24), about 30,000 local pollution control agreements in Japan and over 40 voluntary programmes in the US (OECD 1999), and these numbers were steadily increasing.

Existing research thus suggests that the diversity of EPIs has increased. This thesis will take a slightly different approach and depart from a given *policy area* (in this case, municipal waste policy) at the national level and identify the adoption of instruments, rather than examine the uptake of a certain *type* of EPI (e.g. a carbon tax or a VA on energy efficiency savings) across environmental policy areas or across a large set of countries. Through categorising instruments, the aim is to clarify the diversity patterns in the Swedish and English municipal waste policy mixes and their evolution. Have more so-called 'new' EPIs been adopted, and, if so, which types, when and for what? And what are the patterns regarding the level of coerciveness of instruments added to a mix; have EPIs become increasingly 'harder' or 'softer'? Finally, how do the patterns in the field of municipal waste policy resonate with the wider national EPI mix? A theoretical background for addressing these questions will be provided in Chapter 2.

1.4 Increasing attention to procedural rationality in EPI choice

As described above, the second expectation arising from the quest for new EPIs is that the policy *process* should be conducted in such a way that a comprehensive range of possible instruments – including new kinds of instruments – are compared and assessed in terms of their potential to reach stated objectives. Intentions to take such measures, and consequently approach the ideal of procedural rationality in a decision-making process, have also been stated by policy-makers at the international and national level.

Agenda 21 (UNCED 1992) set out an ambitious programme on ways to improve decision-making processes related to environment and development issues:

> "(a) Improving the use of data and information at all stages of planning and management, making systematic and simultaneous use of social, economic, developmental, ecological and environmental data…
> (b) Adopting comprehensive analytical procedures for prior and simultaneous assessment of the impacts of decisions, including the impacts within and among the economic, social and environmental spheres… analysis should also include assessment of costs, benefits and risks;
> (c) Adopting flexible and integrative planning approaches that allow the consideration of multiple goals and enable adjustment of changing needs…
> (f) Using policy instruments (legal/regulatory and economic) as a tool for planning and management, seeking incorporation of efficiency criteria in decisions; instruments should be regularly reviewed and adapted to ensure that they continue to be effective…[paragraph 8.5]"

At the EU level, the 2001 *White Paper on European Governance* resulted in more ambitious rules and guidance for how to conduct the legislative process (Commission of the European Communities 2002a), minimum standards for consultation (Commission of the European Communities 2002c), and impact assessment of legislative and other policy proposals (Commission of the European Communities 2002b). The guidance on impact assessment (renewed in 2005) prescribes a typical ends-means decision-making process strongly founded upon the ideal of procedural rationality: assessing the problem;

clarifying the objectives; considering alternative policy options; assessing economic, social and environmental impacts; and comparing the options' impacts and ranking them (Commission of the European Communities 2005a, pp. 16-48). For the stage of considering alternative options to traditional regulation, instruments such as self-regulation and economic incentives are mentioned (p. 23).

In the UK, there has been a similar trend of emphasising the need to ensure procedural rationality in the policy process in the 1990s, through procedures such as Compliance Cost Assessments, Risk Assessments, and Regulatory Appraisal (see Persson 2003, Appendix, p. i). In 1998 they were replaced by a single procedure or tool, namely Regulatory Impact Assessment (RIA). In conjunction to the 1999 *Modernising Government* white paper, a report setting out a model of 'professional policy-making' was also published by the Cabinet Office, in which the message of 'policy design based on outcomes' was strongly conveyed (Cabinet Office 1999). In terms of procedural requirements, there has thus been a considerable effort to improve the guidance for and practice of RIAs at central government level (Regulatory Impact Unit 2003; National Audit Office 2006b). A key feature in both the *Professional policy-making* report and the RIA procedure is to define ends prior to means, and consider a wide range of alternative instruments. The Environment Agency has also applied these thoughts more specifically to environmental policy and regulation in its 'model of modern regulation' (Environment Agency 2003). A process is described whereby desired outcomes should first be defined, followed by a consideration of alternative instruments. The Agency states that "[o]nce we are clear about the outcomes we are aiming for and the risks involved, we can look for the most effective approach to achieve the environmental objective with the most efficient use of resources" (p. 12). More recently, the UK has been praised by the OECD for its system of RIA (OECD 2002b pp. 30, 45-46), and the National Audit Office has also noted improvements in RIA compliance and practice in recent years (National Audit Office 2006b).

In Sweden, the overall policy style has been characterised as traditionally 'rationalistic' (see Ruin 1982, Lundqvist 2004), based on the strong belief in the use of knowledge, the more anticipatory to analysing policy problems, and the open and consensual style of deliberation. Presumably, this policy style would be more rather than less conducive to procedural rationality. Also in Sweden there are formal ways of introducing elements of procedural rationality in the policy instrument choice process, however. The need to consider alternative policy instruments has been recognised since the 1970s, when the first requirements for RIA (*konsekvensutredning*) were made (Riksrevisionsverket 1996). Some form of RIA is now required in all stages of the Swedish policy process; at the committee of inquiry level[4] where policy issues are analysed and strategic proposals are made (Statsrådsberedningen 2000), at ministerial level where legislative proposals are drafted (Regeringens skrivelse 1997, pp. 71-72), and at the central government authority level where secondary legislation is prepared (Ekonomistyrningsverket 2003). The Swedish EPA (SEPA) has also more recently provided specific guidance on how alternative instruments should be considered in the environmental policy process (Naturvårdsverket 1999b; Naturvårdsverket 1999a). Compared to the UK, however, the practice of RIA lags behind according to the OECD (2007 p. 14).

Both internationally and at the national level in the UK and Sweden there has thus been a tendency to increase attention and commitment to the strengthening of procedural rationality in the policy process, in order to facilitate comprehensive consideration alternative EPIs, including new types. As explained above, this is in stark contrast to the academic literature which has generally discarded the ideal of procedural rationality as a plausible model for policy-making. A 'best-tool-for-the-job' rhetoric has thus not only been expressed in instrument choice *outcome* terms, but also with reference to the *process* for choosing instruments. The key question to be explored in this thesis is to what extent this ideal of procedural rationality has been achieved in practice and how it has influenced the consideration of alternative EPIs? If not, what are the main barriers? Could it ever be something more than *post-hoc* rationalisation? The conclusion of the thesis will also address the question of the relationship between increased procedural rationality in the choice process and decisions to use NEPIs to a higher degree. Besides investigating the achievement of procedural rationality, other explanations of the nature of the instrument choice process need to be considered. The literature on policy instrument choice offers a rich palette of theories, ranging from the policy style and political culture at the macro-level, to the role of policy networks and ideas at the meso-

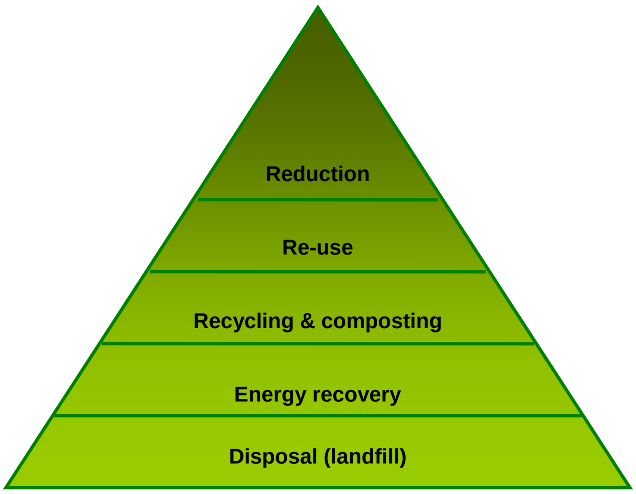
[4] In Sweden, committees of inquiry are appointed by the government to examine issues of current interest. The committees gather relevant knowledge related to a problem and may make a policy proposal. The inquiry reports are normally followed by a consultation round and, if the government chooses, a government bill presented to the parliament.

level, to the bargaining between interest groups at the micro-level. This literature will be reviewed in chapter 3.

1.5 Case studies of municipal waste policy in Sweden and England

The research questions of this thesis relate to large and complex debates on trends within contemporary environmental policy. In order to investigate them in a methodologically robust way, an exploratory research design was developed that focuses on one environmental policy field and that contrasts two European countries. *Municipal waste policy* is particularly interesting and relevant to study in relation to the quest for new and more diverse EPIs, not only because it appears to be currently less researched from an instrument mix and choice process point of view than, for example, climate policy (see e.g. Sorrell and Sijm 2003; Holm-Pedersen 2003; Smith 2004) and agri-environmental policy (see e.g. Daugbjerg 1998ab, 1999). While municipal waste has been an issue on the environmental policy agenda for a long time, the problems of increasing generation and increasing hazardousness have not been solved. It has been estimated that *per capita* municipal waste generation nearly doubled between 1985 and 2000, from about 300 kg to 540 kg in selected EU countries (Commission of the European Communities 2003). The OECD has estimated that it will increase further to 640 kg per capita per year in 2020 in the OECD countries, based on the current trends towards smaller households and increasing consumption and affluence levels (OECD 2001b, p. 236). This suggests that policy instruments used so far have not been sufficiently effective in limiting waste generation. While reducing waste generation is the highest level in the commonly accepted 'waste management hierarchy' (see **Figure 1**), there are also concerns that advancement is too slow at lower levels. In the EU, landfill disposal remains a dominant waste management method (Eurostat 2005, p. 40), and there are also concerns that incineration (with energy recovery) is expanding too fast at the expense of increased material recycling and composting.

Figure 1. The waste management hierarchy



Source: Adapted from Strategy Unit (2002, p. 9).

To address the growing problems of municipal waste management, several demanding pieces of new legislation and strategic objectives were adopted at the EU level in the 1990s and early 2000s (see Haigh 2002; Porter 1998). Therefore, it is now important to study how these requirements and objectives have been translated into concrete policy instruments at member state level.

Policy-makers have indeed responded to this challenge of increasing municipal waste by calling for new forms of EPIs on a wide European basis, to address the wide range of stakeholders involved in waste management (households, local government, waste management industry, etc.). In the work on the recent EU *Thematic Strategy on Prevention and Recycling of Waste*, the Commission stated that "although traditional regulatory measures can play a role [in improving *waste prevention*], they are rarely effective in isolation in such a complex context… This requires policy approaches which go beyond waste policy in its strict sense and enter into fields such as resource management and Integrated Product Policy" (Commission of the European Communities 2003, p. 28, my italics). To promote *waste recycling*, the use of economic and market-based instruments was considered to be "the most promising way" (p. 30). More widespread use of landfill taxes, producer responsibility schemes, tradable certificates and pay-as-you-throw schemes was proposed, as well as incentive systems (e.g. participation in environmental management system schemes) and prescriptive instruments (e.g. landfill bans). However, a review of EU waste directives has shown that few new types of EPIs have been developed within the realm of EU policy so far, suggesting a divergence between policy intentions and practice (Rittberger and Richardson 2003). At the national level, on the other hand, the 2001 OECD Environmental Outlook noted that the range of new forms of EPIs adopted for municipal waste is fairly wide, as illustrated in **Table 1** (OECD 2001b, ch. 25), although instrument choice was not studied systematically in this report. A more systematic study of instrument choice within municipal waste policy would thus shed light on whether the quest for 'new' and more diverse EPIs has been more successful at the national level.

Table 1. Instruments used in the municipal waste policy mixes in OECD countries

Economic instruments	Regulatory instruments	Voluntary agreements	Information and other instruments
- User charges on MSW collection - Unit pricing, or 'pay-as-you-throw' schemes - Product charges on packaging, batteries, tyres and home appliances - Positive economic incentives for waste reduction or recycling, such as deposit-refund schemes, payments for goods delivered for recovery, or recycling credits	- Targets for the recovery or recycling of given proportion of MSW - Extended producer responsibility	- VAs for recycling of packaging wastes	- Product information and labelling programmes - Education and consumer campaigns for household recycling

Source: After OECD (2001b, pp. 304-306).

As mentioned above, the strategy chosen to investigate the first research question of this thesis is to identify in a complete and exhaustive way the policy instrument mix in a particular environmental policy field, namely municipal waste, as opposed to examine the uptake of NEPIs (or a particular kind of NEPI) more broadly across environmental policy. This limitation was made for a practical reason, in that a narrower focus allows a higher degree of detail when identifying instruments. More importantly, though, this methodology seeks to identify *all* instruments, both 'old' and 'new', so as to understand the role of potential NEPIs in context and thereby avoid either over- or understating their significance.

In order to generalise the findings from waste instrument mixes to trends in environmental policy as a whole, it would be necessary to somehow evaluate the representativity of waste policy as a policy field. One aspect of representativity will be discussed in the methodology chapter (chapter 4, section 4.2.2), namely the problem characteristics of waste and their implications for what kinds of EPIs are practically feasible. However, a more modest strategy to discern broader trends has been chosen in this thesis. The study of the waste instrument mix is complemented with an overview of instruments in environmental policy as a whole – at a lower level of detail than the study of the waste instrument mix – and the trends found are then compared.

This study thus focuses on municipal waste policy, while an overview of instrument choice within environmental policy as a whole will be presented as well. It should also be emphasised that this research is limited to examining municipal waste policy, not policy instruments for industrial, agricultural or commercial waste, nor hazardous waste[5]. The main focus is on policy instruments that have been introduced for moving municipal waste management up the waste hierarchy, which was adopted by the EU in 1989 and constitutes a basic policy principle accepted in both Sweden and the UK, and not instruments concerned with very specific waste streams. Furthermore, the research is delimited to policy instruments adopted at the national level, and does not consider local policies and instruments. However, the focus on the national level also means that the impact of the rapidly increasing body of EU waste policy and legislation in the 1990s and early 2000s will be considered.

With regards to the choice of countries, the existing literature on EPI choice has both compared multiple European countries (e.g. Jordan et al. 2003c; Golub 1998a) and studied the development over time in a single country (e.g. Macdonald 2001; Cook 2002). Studying EPI choice *processes* in detail requires an in-depth understanding of the national policy context, in terms of policy style, actor networks, policy paradigms, etc. Considering the process-oriented second research question of this thesis, studying multiple countries within the EU was not possible within the scope of this thesis. However, in order to contrast and highlight different aspects of the EPI choice process and its degree of procedural rationality, a two-country research design was chosen, rather than a single-country design (cf. Daugbjerg 1998b; Howlett 2000a). The

[5] Eurostat's definition of municipal waste is used in this thesis: "Municipal waste includes household waste and similar waste. The definition also includes: bulky waste (e.g. white goods, old furniture, mattresses); and yard waste, leaves, grass clippings, street sweepings, the content of litter containers, and market cleansing waste, if managed as waste" (Eurostat 2005, p. 33). Household and similar waste is defined as "waste from households as well as other waste, which, because of its nature or composition, is similar to waste from households" (*ibid*.).

comparative politics literature explains that a two-country study "forces greater specificity on the researcher" with respect to determining the influence of various factors on an outcome than a single-country study (Peters 1998, p. 4). Note that, since the role of procedural rationality for EPI choice has not been systematically studied before, the analytical aim of this study is to *explore* issues, in order to subsequently refine the methodology and conduct more country studies. The aim is not to strictly test hypotheses or make a 'causal-analytic' comparison (Ragin 1987 pp. 36-42; Hopkin 2002 p. 250), with the aim to provide generalisable results for other European countries.

Nevertheless, justification of the country selection and the control of 'extraneous sources of variance' (Peters 1998, p. 33) is required also for more exploratory studies. The starting point for the selection in this study is that the two countries should, over the past decade, have clearly *stated intentions to achieve a more diverse EPI mix and procedural rationality* in the choice process, i.e. evidence of a 'quest for NEPIs'. This follows from the research objective to take stock of the success in translating rhetorics into practice. Above (section 1.3), it was described how governments in the UK and Sweden have indeed committed to considering a wider range of alternative instruments. In particular, new political momentum was given to the search for NEPIs around the same time in these two countries. In 1996, the new Social Democratic Prime Minister of Sweden, Göran Persson, announced an ambitious new policy agenda for a 'transition to an ecologically sustainable society', inspired by ecological modernisation ideas (Lundqvist 2000; Fudge and Rowe 2001).[6] This prompted an interest in adding new instruments such as large environmental subsidy programmes and management systems to the Swedish repertoire (which was already characterised by a relatively high use of environmental taxes as a NEPI). In 1997, the new Labour government in the UK announced that environmental taxes and other economic instruments were to be used on a wider basis (see HM Treasury 2002; Jordan et al. 2003d p. 188), also in the context of ecological modernisation ideas (Revell 2005 p. 347). A 'most-similar' research design has thus been applied (Hopkin 2002 p. 254), with the basic expectation is that similar results would emerge in Sweden and the UK as regards the success in the quest for NEPIs, and if not, for predictable reasons.

[6] In Sweden, the Social Democrats held office from 1994 to 2006, i.e. throughout the study period in this thesis.

However, more European countries than Sweden and the UK have stated their intentions to diversify EPI repertoires and consider alternative instruments in a procedurally rational way. There is a need for a more specific rationale, which will also shed light on the implications for interpreting the results. One strategy could be to choose countries on the character of the EPI mix and the success in adopting NEPIs. Existing comparative studies of EPI choice at the national level suggest that European countries (i.e. those that have been covered in the studies) could be grouped into three groups: higher use of NEPIs – the Netherlands, Denmark, Finland, Sweden and the UK; medium use of NEPIs – Germany, France, Spain, and Italy; and lower use of NEPIs – Greece and Ireland (Jordan et al. 2003a p. 209, Figure 1; OECD 1999b, 2000ab, 2001def, 2002b, 2003f, 2004cd, 2007).[7,8] From this perspective, Sweden and the UK also appear to be more or less 'similar' cases, together with the Netherlands, Denmark and Finland. Because 'NEPIs' is such an aggregated term, one could compare countries in relation to specific NEPI types. For example, if environmental taxes were in focus a comparison between some of the Nordic countries would be more of 'similar' cases than the UK. However, choosing countries based on NEPI adoption as an outcome variable is altogether conceptually flawed with respect to the research aim of this thesis. It has been stated above that more comprehensive *consideration* of alternative EPIs does not necessarily need to result in the *adoption* of a NEPI. The comparative politics literature has generally advised against such choosing of cases based on certain values of the 'dependent variable' (Peters 1998 p. 31). This strategy could thus produce misleading results.

Instead, countries should here be chosen with respect to the *process* of EPI choice. With the stated intention to consider a wider range of EPIs as the starting point for country selection (see above), the practice of RIA can be used as an indicator of the procedural rationality aspired to within different countries. As described above, consideration of alternative instruments is a key feature of RIA. Again, European countries can be roughly grouped based on existing comparative reviews of RIA practice. The OECD

[7] The OECD references refer to the background report *Goverment Capacity to Assure High Quality Regulation* that has been prepared for the all the OECD country reviews on Regulatory Reform. All the reports on European countries to date have been reviewed here and countries were grouped based on the OECD's assessment of innovation with regards to EPIs.

[8] Note that combining the comparisons of country performance in Jordan et al. (2003a) and the OECD reports suffered from two problems; the country coverage differed and the time of assessment differed. It is still considered that these studies give a sufficient indication of comparative NEPI use.

Regulatory Reform reports (OECD 1999b, 2000ab, 2001def, 2002b, 2003f, 2004cd, 2007) and a report prepared for the EU Directors of Better Regulation Group (Formez 2004) suggest the following groups: first, the UK is seen as a forerunner, followed by the Netherlands, Denmark, Sweden and Finland; second, Germany, France, and Austria are at a less advanced level; and third, Italy, Greece, Spain, and Ireland are assessed as showing greater weaknesses.[9] However, this grouping is less clear-cut than the one regarding NEPI adoption, and different assessments use different performance criteria (cf. for example Hertin et al. 2007).

Out of the countries with better RIA performance Sweden and the UK were chosen here. The main reason is that while both these countries evidently perform relatively well, they have different approaches to RIA, including the consideration of alternative instruments (Persson 2003; Hertin et al. 2007). In the UK, there has been a considerable (top-down) effort to introduce a standardised, compulsory RIA format that requires the identification and assessment of alternative instruments (see Regulatory Impact Unit 2003). In Sweden, on the other hand, there is not a standardised format for RIA at the most strategic level, i.e. in the commissions of inquiry. In Sweden, these commissions are more free regarding the scope and terms of reference of the assessment (see Lundqvist 2004 p. 160ff; Hertin et al. 2007 p. 8). However, contrasting these two different approaches is helpful for exploring issues around procedural rationality in the process and how it relates to the consideration of NEPIs. Can procedural rationality and the consideration of alternative EPIs be achieved in different ways?

In addition, it appears that an empirical comparison between Sweden and the UK can contribute to a current gap in the comparative EPI choice literature. For example, Sweden appears to be less researched in recent EPI choice literature than the Netherlands (see e.g. Zito et al. 2003; Liefferink 1998). Also, several comparisons between the Nordic countries in relation to environmental taxation have already been made (see e.g. Daugbjerg 1998b; Daugbjerg and Svendsen 2001; Holm Pedersen 2003), suggesting that a comparison involving the UK could be informative.

[9] Note that combining the comparisons of country performance in Formez (2004) and the OECD reports suffered from two problems; the country coverage differed and the time of assessment differed. It is still considered that these studies give a sufficient indication of comparative RIA practice.

To sum up, the UK and Sweden are contrasted in this study. In the UK a large part of waste policy is made at the level of the devolved administrations. Therefore, the primary focus in this thesis is on England, although some of the waste policy is common to the whole of the UK. Sweden and the UK (England) were chosen since they both belong to the group of (similar) countries that have stated an intention to consider alternative EPIs to traditional regulation and that have the measure of RIA in place, which should allow for an operationalisation of this intention. With this basic *similarity*, there are also *different* approaches in the two countries which should be useful for exploring issues around how procedural rationality may or may not work in practice, and the reasons for this. However, this choice of countries also has its limitations. A wide range of potential contextual differences between the two countries, such as political culture, party-political system and nature of policy networks, can affect the findings. Some of these contextual differences are likely to be smaller between the Nordic countries. To structure the analysis of such contextual political and institutional factors and their effects, a comprehensive analytical framework is developed in chapter 3.

The aim of this thesis is to study recent changes (or lack thereof) in EPI choice and the *time period* studied is 1995-2005. The delimitation of the time period follows from the research questions. Since the quest for NEPIs gained momentum in the early 1990s with statements such as Agenda 21 (see section 1.3 above; see also chapter 2), the year of 1995 was chosen as a starting year for the study. Of course, a more historical study could reveal other patterns, for example that so-called 'new' EPIs had been tried and tested before the mid-1990s, with the implication that recent uptake of NEPIs could risk being overstated. The country studies in Jordan et al. (2003c) analyse the use of NEPIs from back in the 1970s, while Macdonald (2001) and Howlett (200a) provide more longitudinal studies of EPI use in Canada and the US from the first half of the 20th century and the 1960s respectively. There are two reasons for the shorter time period selected in this study. First, the focus is on the more or less concerted efforts to find new and more diverse instruments for municipal waste policy as a response to new national and EU waste legislation and strategies over the last decade, as opposed to a less intentional evolutionary process. Second, the depth of analysis, in terms of complete and exhaustive identification of instruments, is prioritised over breadth. At the same time, a time period of less than a decade would be excessively short. Reviewing policy change over a decade allows for sufficient time for policy proposals to be finally adopted. EPI choice processes may take a considerable number of years. Furthermore, it also allows for changes in policy-making routines and procedures, such as introducing or consolidating the use of RIA, to take effect. Taking these considerations into account, 1995 was chosen as the starting point. Importantly, this was also the year when Sweden became a member of the EU and thus became subject to the same European waste legislation and strategies as the UK.

To address the first research question, overviews of the national-level policy instruments used for environmental policy as a whole and for municipal waste policy specifically have been made. To address the second research question on the process of instrument choice, one recently adopted EPI in each country has been selected as a case study for analysis of whether they were preceded by systematic comparative assessment of alternative instruments. Importantly, the starting point for case selection here is the *process* of instrument choice, rather than the resulting instrument – whether a NEPI or not. The aim is to study the *consideration* of alternative EPIs in a policy process. Three main considerations were made in the selection of instruments for case study. First, the instruments had to be recently or soon-to-be adopted, in order to allow for more recent changes in routines and procedures in policy-making for better consideration of alternative EPIs (such as RIA) to potentially have taken effect and to capture more or less current practice. Second, for practical reasons, the processes had to be rather discrete in nature, so that a timeline and debate could be traced. Finally, the resulting instruments should preferably be of a significant importance to the municipal waste sector, as opposed to minor instruments or amendments of existing instruments. Although the latter are also worthwhile studying to understand consideration of alternative instrument designs, the former are more likely to involve more strategic considerations of EPI types.

Without an ‘abundance’ of potential cases[10], these criteria led to the selection of one major economic instrument in each country. In England, the *landfill allowance trading*

[10] The following recently introduced instruments were considered for selection, but were not chosen in the end due to the ‘significance’ criterion: *England* – Incineration emission standards (2002), Landfill emission standards (2002), Producer responsibility end-of-life vehicles (2003); *Sweden* – Bans on landfill of combustible and organic waste (2002, 2005), Batteries recycling information campaign (2002). For reference, see Appendix V.

scheme introduced in 2005 was chosen. This is the world's first tradable permits system (TPS) targeted at the public sector, and was adopted to meet the challenging targets for reducing landfill of biodegradable municipal waste that were set in the 1999 EU Landfill Directive. An innovative instrument was thus chosen, and this thesis will examine the reasons and to what extent this choice of EPI was a result of procedural rationality in the process. In Sweden, the *waste incineration tax* introduced in 2006 was chosen as a case study. It represents a 'proactive' instrument not called for by international obligations or concrete national targets, but was instead the result of a lengthy discussion and negotiation spanning more than a decade. To what extent are these two instruments the result of 'typical' cases of EPI selection processes (see Yin 1994)? Whilst it should again be emphasised that this is an exploratory study that does not aim for broad generalisation, there are no *a priori* reasons to assume that these processes are highly atypical or unrepresentative cases. However, various case-specific circumstances that affected the conduct of the EPI choice process, as well as the final EPI type chosen, will be identified and discussed in the results chapters (chapters 7 and 8).

1.6 Conclusion and structure of the thesis

The aim of this thesis is thus to take stock of the success (or lack thereof) of the UK and Sweden in the quest to find new EPIs. In doing so, it will address the broader question of whether the process of EPI choice has become increasingly depoliticised and a 'best-tool-for-the-job' approach has evolved. To complement existing research on particular EPI types (such as green taxes, tradable emission permits, VAs, or eco-labelling schemes) and their increasing use in different countries and sectors, this thesis will instead depart from a specific environmental policy field in two countries, namely municipal waste policy at the national level in the UK and Sweden respectively. The aim is to investigate two interlinked aspects of EPI choice. First, the notion of a '*policy instrument mix*' will be examined, in terms of the diversity of EPIs and whether there is some pattern of increasing or decreasing degrees of coercion of EPIs added to the mix over time. Second, the degree to which the *process of EPI choice* is conducive to consideration of a wide range of potential instruments will be analysed, through case studies of the development of a landfill allowance trading scheme in the UK and a waste incineration tax in Sweden. The question here is to what extent procedural rationality measures characterise the policy process, and to what extent other political and institutional explanations of EPI choice are needed.

To develop the research questions and analytical framework further and to eventually present the empirical case study results, the thesis has been structured in the following way. In *chapter 2*, definitions and typologies of policy instruments proposed in the literature will be reviewed, along with a historical overview of how new instrument types have been gradually added to the arsenal of EPIs. The chapter is concluded by the establishment of an EPI typology to be used in this thesis and a discussion of the emerging interest in policy mixes. The literature review is continued in *chapter 3*, but with a focus on the choice process rather than instruments as outputs. First, the notion of procedural rationality is unpacked. Despite the substantive critique of its plausibility, it is argued that it is worthwhile to study the degree of procedural rationality in practice. However, other explanations of the instrument choice process must also be considered. A macro-, meso- and micro-level framework is used to review alternative political and institutional factors, in which relevant theories are structured according to their emphasis on structural factors or human agency. The literature reviews in chapter 2 and 3 are used to construct the analytical framework for this thesis in *chapter 4*. There are three key parts in the analytical framework; an EPI typology to facilitate the study of mixes, a list of six key operational elements of procedural rationality, and a summary of political and institutional factors explaining EPI choice processes. The methodology, research design and data collection methods used in this study are also described in chapter 4.

The empirical part of the thesis begins with a background study of waste policy in Sweden and the UK in *chapter 5*. This chapter provides a brief historical overview of waste policy development, including the growing influence of EU waste legislation at the national level. The key waste policy objectives and targets for which instruments have been devised are reviewed. *Chapter 6* then begins with an overview of the mix of policy instruments for environmental policy as a whole and its evolution in the two countries over the last decade. A more detailed examination of the municipal waste policy instrument mixes follows. The mixes in Sweden and England are systematically compared and the existence of particular patterns of adoption is evaluated. The relationship between coerciveness and effectiveness is discussed, and the chapter is concluded by an overview of general use of the various instrument categories in environmental policy as a whole. *Chapters 7* and *8*, then, present the results of the case studies on the two policy processes selected. They begin with a description of the instrument finally adopted, the course of the process, the institutional context and main stakeholders involved. The extent to which procedural rationality was achieved is then assessed, through the framework of six key elements developed in chapter 4. Finally, the plausibility of the other instrument choice theories in the macro-meso-micro level framework developed in chapter 4 are discussed. The results from the mix and process studies are then discussed and evaluated in *chapter 9*. Lastly, in this chapter the key results are summarised, the conclusions and policy implications are presented, and needs for future research are identified.

Chapter 2

Instruments for Environmental Policy: From Traditional Regulation to a Diverse Mix

2.1 Introduction

In the previous chapter, commitments of the Swedish and UK governments to use a diverse mix of EPIs, including economic instruments, VAs, information measures and self-regulation, were quoted. These stated policy intentions led to the formulation of the first research question, namely; *has the EPI mix become more diverse over time?* Have the Swedish and UK policy-makers delivered on their commitments? In order to address this question, there is a need to define and describe the study object; policy instruments. The literature review presented in this chapter will start with the basic conceptual building blocks for studying instrument choice, i.e. a definition of policy instruments and an instrument typology. With the variety of EPIs clarified, the empirical literature on EPI use is briefly reviewed. The emergence of new kinds of EPIs is described, followed by a discussion of causes and drivers of this shift. It is concluded that there has recently been a shift from focusing on single kinds of EPIs to recognising the need for comprehensive mixes, arguably to the point where high instrument diversity is seen as an end in itself.

2.2 Defining policy instruments

While much effort has been invested in constructing EPI typologies (see next section), general definitions of EPIs are conspicuously absent. Also in the general public policy literature, it has been argued there is a lack of coherence in the literature on public policy instruments regarding definitions (de Bruijn and Hufen 1998, p. 13). However, most conceptions seem to subscribe to some version of an ‘ends-means’ model of procedurally rational decision-making, in which (i) a policy objective is set before the instrument is chosen, (ii) the instrument chosen represents the alternative that is most optimal or appropriate according to given criteria, and (iii) the objective and instrument are analytically separable entities. Howlett's definition of policy instruments is representative; "the myriad techniques at the disposal of governments to implement their public policy objectives" (Howlett 1991, p. 2).[11]

A couple of issues have been raised in the literature that problematise this kind of definition, though. First, de Bruijn and Hufen (1998, pp. 13-14) have proposed that 'techniques' should be limited to including instruments as *objects* and not as *activities* (e.g. bribery, rhetoric, 'sweet talking'). They argue that including activities would blur the line between the policy as a whole and the actual, specific instrument to implement it. 'Organisation' – an "ensemble of acitivities and processes" (*ibid*., p. 14) – is sometimes seen as an instrument (see e.g. Hood 1986). However, Vedung (1998, p. 38) argues that it should be seen as "a prerequisite for the application of policy instruments" rather than a policy instrument in itself, or as a broader public governance strategy rather than a policy instrument in the narrow sense.

Second, the definition proposed by Vedung (1998) challenges the rather power-neutral definition by Howlett. He defines policy instruments as "the set of techniques by which governmental authorities wield their power in attempting to ensure support and effect social change" (p. 21). First, this definition makes clear that instruments are not void of 'power wielding', i.e. they are not purely instrumental or mechanic but create, modify or reinforce a certain power relationship between the government and other actors. Second, Vedung's definition suggests that the purpose of an instrument does not need to be confined to achieving a given substantive policy objective ('effecting a certain social change'), but it can also be chosen to ensure (continued) political support.

Another, less explored question concerns the classification of *policy targets* as instruments. It has been observed that "[m]ore recently, targets and indicators have been integrated into the arsenal of policy instruments" (Ministry of Environment 2002, p. 113). The culture of target-setting under the UK New Labour government is widely recognised (see e.g. Anon. 2003a). Also Sweden has put a lot of effort into developing a comprehensive system of national environmental objectives, targets and indicators (see

[11] Note that a range of alternative terms to 'instruments' are used in the literature, such as 'tools', 'means', 'options', and 'techniques'.

Regeringens proposition 2001, Lundqvist 2004). However, while targets and indicators may have come to occupy a more central – or just more visible – space in environmental policy, it may be premature to think of them as instruments until there is more research on their role in contemporary policy. First, to the extent that they apply to external, non-governmental actors, it could be argued that they should be considered as part of (non-binding) voluntary agreements (VAs) or (binding) regulation. Second, the notion of 'target as instrument' is completely contradictory to an ends-means view of policy instruments. On the one hand, it could be argued that there is in fact a hierarchy of policy objectives and that a concrete, quantified target could be seen as a tool for steering behaviour towards a higher objective (e.g. 'reduce CO2 emissions') through giving an actor a more manageable task (e.g. 'increase the proportion of green electricity of your total electricity consumption by X % within Y years'). On the other hand, it could be argued that the actual instrument here is in fact persuasion or coercion, for example by threatening with sanctions or future binding regulation. In any case, it seems appropriate at this stage to think of this phenomenon not as 'targets as instruments' but as 'targets without instruments'.

As a working definition in this thesis, a slightly modified version of Vedung's definition above is proposed: *the set of techniques by which governmental authorities wield their power in attempting to ensure support and achieve public policy objectives*. Although the focus will be on stated substantive objectives, it highlights the potential existence of informal and indirect objectives (such as ensuring voter or interest group support) that need to be considered if not exhaustively researched. Furthermore, it is proposed here that 'activities' should not be seen as instruments and that policy targets should be seen as part of VAs or regulations or as 'targets without instruments'.

2.3 Categories and typologies of EPIs

Much work has been done in the literature to develop typologies, both for academic purposes and because it has been seen as 'crucial' for policy-makers to have a good overview of the generic forms of instruments (Vedung 1998, p. 21)[12]. However, this

[12] For an example of typology orientated towards supporting policy-makers, see the 'policy selection matrix' developed by Sterner (2003, pp. 214-215, table 18.1) that lists six types of EPIs and their performance on fourteen criteria and conditions (such as static efficiency, complexity, distribution and political issues, rent-seeking).

section will primarily bring attention to the *variety* of EPIs, with the aim of establishing a working typology for this study. It will not describe particular types of EPIs or contribute to the debate on the role or function of typologies as such[13].

Some studies simply categorise EPIs, without typifying them further in relation to some parameter. For example, the OECD has identified six general categories as mentioned in the previous chapter: command-and-control instruments; economic instruments; liability, damage compensation; education and information; voluntary approaches; and management and planning (OECD 2001c, p. 132). For a comprehensive overview of similar EPI categorisations by different authors, see Richards (2000, Table A2). An example of a slightly more elaborate typology is that provided by the DEFRA Regulation Taskforce 2004 (see **Table 2**). Here, instruments are grouped into four main groups and the market failure(s) each type addresses is specified.

Table 2. DEFRA Regulation Taskforce's typology of regulation

Type of instrument	Market failure addressed	Example
Softer instruments		
Suasion	Incomplete information, externalities, merit goods	Kerbside household recycling schemes
Information provision	Incomplete information	Food labelling
Self-regulation or voluntary agreement	Incomplete information, externalities	Management of environmental impact of mineral extraction
Voluntary management systems	Incomplete information, co-ordination failure	Accredited environmental management systems
Negotiated agreement	Incomplete information, externalities	Reduction of ozone depleting chemicals
Market-based instruments		
Tradeable permit or quota	Externalities	Carbon emissions
Liability	Externalities	Oil spills
Tax or charge	Incomplete information, externatlities	Road congestion
Market creation through facilitated bargaining between parties or payments for environmental services	Externality	Direct payments for conservation – e.g. agri-environment schemes
Prescriptive instruments		
Environmental standard	Externalities	Consents for discharge to surface waters
Performance standard	Externalities	Vehicle emission standards
Process specification	Externalitites, merit goods	Regulation of radioactive substances
Economic regulation		
Subsidy	Intellectual property rights, merit goods, public goods	University research centres; energy efficiency subsidy for poor households
Economic regulation	Monopoly	Water industry

Source: DEFRA Regulation Taskforce (2004, Annex C).

[13] For theoretical discussions of the nature and functions of typologies, see Linder and Peters (1989), Howlett (1991), Howlett and Ramesh (1993), and Vedung (1998).

Daugbjerg (1999, p. 168) argues that categories such as these are not sufficient; "[i]f we assume that the choice of instruments is political, the typology must express the political properties of the instruments". The more deductive typologies in the literature on public policy instrument instruments address these. In 1971, Anderson (1971, p. 129) identified four basic 'forms of public policy' available to the state: deployment of political attributes (i) as monopoly of legitimate force and (ii) as a focus of authority and prestige, and, as an economic institution, (iii) derivation of resources from the society through taxation, borrowing and sale and (iv) spending for public purposes. Hood (1986, pp. 124-125) uses a similar approach and identifies four basic resources the government can draw on; nodality, treasure, authority, and organisation. Depending on the purpose of public policy – to detect or effect behaviour – there are eight different kinds of instruments available to the government, see **Table 3**.

Table 3. Hood's model of the tools of government

	Nodality (information)	Treasure (finance)	Authority	Organisation
Effectors	Advice	Grants, loans	Laws	Service delivery
Detectors	Surveys	Consultants	Registration	Statistics

Source: Hood (1986).

Another instrument typology has been developed that focuses on how the government can use its resources to either specify the goal to be achieved or how the goal should be achieved, or both. Jordan, Wurzel et al. (2003b), based on Russell and Powell (1996), have categorised EPIs according to such a scheme (see **Table 4**). This kind of typology is politically orientated in that it addresses the level of freedom for target actors to choose how to behave.

Table 4. Jordan et al's typology of policy instruments

	Regulator specifies the *goal* to be achieved	**Regulator does *not* specify the *goal* to be achieved**
Regulator specifies *how* the goal is to be achieved	Command and control regulation	Technology-based regulatory standards
Regulator does *not* specify *how* goal is to be achieved	Most negotiated VAs; some market-based instruments; some regulation (e.g. environmental quality objectives)	Most market-based instruments; some VAs; informational devices

Source: Jordan, Wurzel et al. (2003b, p. 9).

According to Howlett (1991, pp. 3-4), Hood's typology above is an example of the 'resource approach', which categorises instruments according to the nature of the governing resource employed. The 'continuum approach', on the other hand, categorises according to some choice governments must make. Examples of such continuous parameters that have been proposed include the 'level of intrusiveness' (Linder and Peters 1989, p. 56) and 'nature of instrument membership' (Dahl and Lindblom 1953).

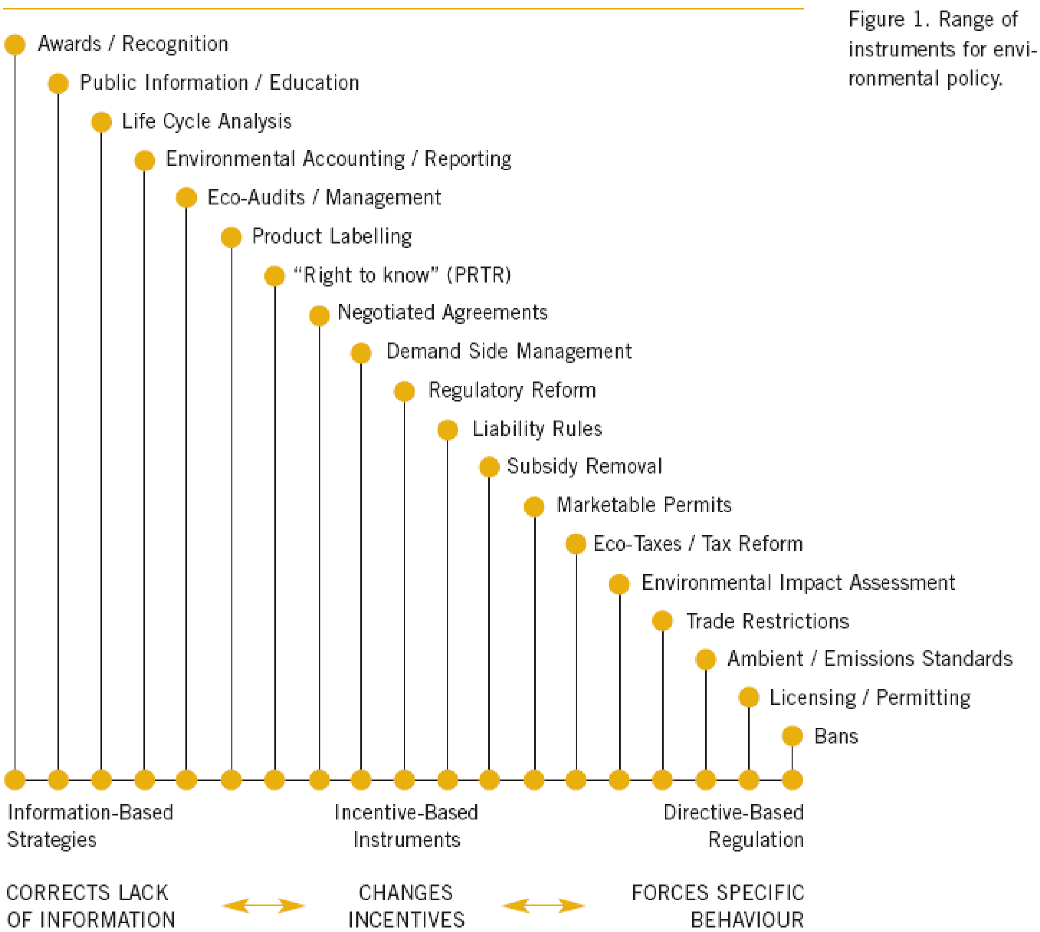
Doern and Phidd (1983) use the 'continuum approach' but reduce the relevant parameter to just one: 'degrees of legitimate coercion'. Minimum coercion is represented by self-regulation and maximum by public ownership, and in between we find instruments involving exhortation, expenditure and regulation (see **Table 5**). Indeed, the level of coerciveness seems to be the central parameter or organising feature in many typologies. The common tripartite typology is often, but not always, used as an indicator of the degree of coerciveness: *regulatory instruments* (coercive, 'sticks'), *financial incentives* (limited coercion, 'carrots'), and *information transfer* (non-coercive, force of conviction, 'sermons') (see e.g. Vedung 1998, de Bruijn and Hufen 1998). An application of this kind of typology to *environmental* policy instruments is illustrated in **Figure 2**.

Table 5. Doern and Phidd's typology of instruments

Private behaviour	**Exhortation**	**Expenditure**	**Regulation**	**Public ownership**
Self-regulation	Speeches Conferences Advisory investigations	Grants Subsidies Transfers	Taxes Tariffs Fines Imprisonment	Crown-corporations Mixed-corporations
Minimum------------------------------Degree of legitimate coercion------------------------------Maximum				

Source: Adapted from Doern and Phidd (1983, p. 111, figure 5.1).

Figure 2. Barde's range of instruments for environmental policy



Source: Barde (2002, p. 115).

Some problems with assuming that the level of coerciveness is the defining feature of policy instruments have been pointed out, however. First, it has been questioned whether it is indeed the most central parameter to the actors involved (Macdonald 2001). Doern and Phidd's account has been criticised for being too normative, in the sense of stipulating a liberal ideal of minimal state intervention that may not be generally accepted (Howlett 1991, Linder and Peters 1989). Second, Macdonald (2001) has argued that the way generic instrument types are designed, implemented and enforced determines the *actual* level of coerciveness, so that instrument types *per se* should not be associated with certain degrees of coerciveness. **Table 6** demonstrates how different EPI types can incorporate both coercive and non-coercive variants. Hence, it could be misleading to assume that policy actors necessarily associate a certain EPI *type* with a certain degree of coerciveness. For example, actors who would generally prefer a low level of state interference may prefer a law, that will not be effectively enforced, over a voluntary agreement, adopted under the threat of future legal or financial instruments. The need to examine the actual, detailed *design* of EPIs – such as the scope of the target group, exemptions and rebates – rather than limiting the analysis to labelling instruments according to type is also acknowledged by Daugbjerg and Svendsen (2001, p. 81) in their comparative analysis of green taxes in the Nordic countries.

Table 6. Macdonald's four categories of environmental instruments

Provide direct environmental service	
Non-coercive	In competition with similar private services
Coercive	As a monopoly, with legal requirements that it be used by all polluters
Encourage voluntary behaviour change	
Non-coercive	As self-regulation, with no accompanying context of other instruments
Coercive	Within a context of other service, legal or financial instruments
Positive or negative financial incentives	
Non-coercive	Not sufficient to significantly influence behaviour
Coercive	Sufficient to significantly influence behaviour
Law	
Non-coercive	Little effort to enforce, through inspection and prosecution
Coercive	Used with an active enforcement program

Source: Macdonald (2001, p. 170).

A similar identification of coercive (‘repressive’) and non-coercive (‘stimulative’) variants of the three core instrument types has been made by van der Doelen (1998) (see **Table 7**). For example, within the ‘economic control model’, the financial incentives provided could be either positive (a subsidy) or negative (a levy).

Table 7. Van der Doelen's stimulative/repressive typology

	Stimulative	**Repressive**
Communicative control model	Information	Propaganda
Economic control model	Subsidy	Levy
Judicial control model	Contract/covenant	Order/prohibition

Source: van der Doelen (1998, p. 133, figure 5.1).

A third and related point is that it is not clear how best to operationalise it into measurable variables. Both Linder and Peters (1989) and Daugbjerg (1999, p. 166) emphasise the need to focus on actors’ subjective perceptions of instruments, rather than to expect instruments to be characterised by actors in an objective way. However, regardless of whether a perceptional approach is taken or not, the literature clearly suggests that there is a need to unpack the concept of coerciveness and understand how it may vary *within* a certain category of EPIs. This issue will be explored in chapter 6, after the discussion of empirical results.

In summary, this review shows that typology construction is a major field of research itself. Typologies can be constructed in several ways, e.g. in relation to the market failures addressed, government resources used, whether a goal and/or means is specified, and the relative level of coerciveness. For this study, we primarily need a list of possible EPIs for comparison with the actual instrument choices made, both in the waste policy mixes and the specific policy processes studied. Therefore, the inductive and empirically-based list of EPI categories provided by the OECD seems useful and straightforward: *command-and-control instruments; economic instruments; liability, damage compensation; education and information; voluntary approaches;* and *management and planning* (OECD 2001c, p. 132).

2.4 Three waves of new environmental policy instruments

The typologies and categorisation schemes propose that – in theory at least – a wide, and expanding, range of possible EPIs are available to policy-makers. What trends in the use of different types of EPIs can be empirically observed? In the European context, three distinct, but overlapping, phases or waves can be discerned in the EPI debate (see Tietenberg 1998, Barde 2002). The first phase was actually characterised by a lack of debate, or a largely unquestioned consensus that traditional 'command-and-control' regulation was the instrument of choice. 'Command-and-control' regulation (henceforth also referred to simply as regulation) had been a strongly dominant and 'default' policy tool since the emergence of national environmental policy institutions and legislation in the 1960s. However, towards the end of the 1970s and in the early 1980s the strong reliance on regulation was increasingly questioned on the grounds of excessive implementation and enforcement costs, lack of effectiveness associated with the 'end-of-pipe' approach, and failure to provide dynamic incentives (Golub 1998b).

A second wave thus started in the 1980s with the growing dissatisfaction with regulation and increased attention to economic, or 'market-based', instruments[14] such as pollution charges, environmental taxes, subsidies, tradeable permit systems and deposit-refund systems. Although several types of economic instruments, for example water use

[14] Note that some authors distinguish between economic and market-based instruments. Here they are used interchangeably.

charges and tradeable air emission quotas, dated back to the 1970s (EEA 2001), there was now a more targeted effort to find such alternatives to regulation and introduce them on a wider basis. For example, the topic of green tax reform was increasingly examined, both theoretically and how it could be practically implemented. This phase in the debate included contributions such as *Blueprint for a Green Economy* by Pearce, Markandya et al. (1989), which was an important milestone in the growing use of environmental economics as a knowledge basis for policy-making. It advocated that "pollution charges are generally better than 'command-and-control'" (p. 162). A few years later, Agenda 21, one of the 1992 Rio 'Earth Summit' outcomes, also addressed the need for increased use of economic instruments;

> "[e]nvironmental law and regulation are important but cannot alone be expected to deal with the problems of environment and development. Prices, markets and governmental fiscal and economic policies also play a complementary role in shaping attitudes and behaviour towards the environment" (UNCED 1992, chapter 8, paragraph 8.27).

The same message was conveyed in the Fifth Environmental Action Programme of the EU, adopted in 1992, and the Maastricht Treaty, which 'heralded a new era' of searching for more flexible and efficient instruments, also at the supranational level (Golub 1998b, p. 1). Finally, the OECD's policy analyses also played a role in promoting the spread of economic instruments for environmental policy.

While economic instruments have continued to be explored, advocated and implemented and traditional regulation has continued to be 'reformed' and 'modernised', a third wave can be discerned from the mid-1990s. Increasingly, systematic attention was given to 'softer' instruments, such as different types of voluntary approaches (VAs), environmental management systems, and information measures such as eco-labels (see e.g. EEA 1997, OECD 1999, Steinzor 1998; Jordan, Wurzel et al. 2005a). According to de Bruijn and Hufen (1998, p. 18), the growing significance of various instruments for 'information transfer' was based on the idea that the 'force of conviction' rather than 'coercion' should be the factor that provides direction to policy. Jordan, Wurzel et al. (2003b) trace the increasing adoption of 'new EPIs' (NEPIs) – i.e. economic instruments, VAs, and informational devices such as eco-labels – to the presumed turn from government to 'governance' (see also Fiorino 1999 and Lafferty 2004). In such a context, the government can no longer act unilaterally and the idea of NEPIs "fitted with the debate about governance – the idea that the state should not seek to interfere in every facet of social and economic life" (Jordan, Wurzel et al. 2003b, p. 14).

Has the attention in the policy debate to new types of EPIs been accompanied by a real change in policy practice? While measuring the use of EPIs and their relative significance in a meaningful way is difficult (and will be discussed in methodological terms in chapter 4), some statistics on the increasing number of and revenues from environmental taxes and the spread of VAs were quoted in chapter 1 (see section 1.3). It was seen that the number of environmental taxes, for example, has increased more than tenfold since the late 1980s (although the level of revenues in relation to total tax revenue has been stable around 7% on average in OECD countries). Also the use of VAs has increased significantly over the last decade. However, there is disagreement in the EPI literature whether this uptake of new alternatives to regulation should be seen as ‘stunningly fast’ (Jordan, Wurzel et al. 2003a) and characterised by ‘virus-like’ (Rittberger and Richardson 2003) and ‘rapid diffusion’ (Tews, Busch et al. 2003, Busch, Jörgens et al. 2005) across national borders, or whether it has been slower than expected (Hockenstein, Stavins et al. 1997) or even occurring at ‘glacial pace’ (Dryzek 1997, p. 116).

There is also a lack of comprehensive assessments of the international variation in the use of NEPIs[15]. However, the recent overview conducted by Jordan, Wurzel et al. (2003a) showed that there are significantly different patterns of NEPI adoption in European countries (and Australia) (see **Table 8**). The team of researchers found that NEPIs have almost become the preferred types of instruments in the UK, Finland and Germany, while they are barely used at all in Ireland. These results are explained by the importance of national institutions in filtering out new ideas about EPIs, which has meant that pre-existing national repertoires have survived to a considerable extent (see also Tews, Busch et al. 2003, p. 576).

[15] A review of existing sources will be made in chapter 6, with a specific focus on Sweden and the UK. For European country case studies on the use of NEPIs, see Golub (1998a) and Jordan, Wurzel et al. (2003c). For a study of the spread of a selection of specific NEPIs (e.g. carbon tax, national strategy for sustainable development) in OECD countries, see Tews, Busch et al. (2003).

Table 8. Pattern of NEPI adoption

	Ecotaxes	Tradeable permits	Voluntary agreement	Eco-labels	Regulation
Australia	Low	Low	Low	Low	Still dominant
Austria	Medium	Low	Low/medium	Medium	Still dominant
Finland	High	Low	Medium	High	Still significant
France	Medium	Low	Low	High	Still dominant
Germany	Medium	Low	High	High	Still dominant
Ireland	Low	Low	Low/medium	Low	Still dominant
Netherlands	High	Medium	High	Low	Still significant
UK	Medium	High	Medium	Low/medium	Still significant

Source: Jordan, Wurzel et al. (2003a, p. 209).

There is thus a lack of detailed studies examining EPI use, especially where Sweden and the UK are compared and where a particular policy area is in focus rather than a particular type of EPI. By taking an evolutionary approach to the diversity policy instrument mixes used for municipal waste policy in Sweden and England, this thesis will address this gap in the literature.

While regulation may thus still play a significant or dominant role, what were the main drivers behind this gradual extension of the range of available EPIs? Some general factors explaining the extended repertoire of available EPIs have been identified (see Golub 1998b; Fiorino 1999; OECD 2001c; Jordan, Wurzel et al. 2003a; Lafferty 2004; Jordan, Wurzel et al. 2005b; Jordan, Wurzel et al. 2005c), even though rationales for the use of specific types of NEPIs in specific contexts can be significantly different. First, as noted above, there was dissatisfaction with traditional regulation, especially the lack of flexibility and static and dynamic efficiency effects. Furthermore, regulation was seen as too reactive to deal with sustainable development concerns (as opposed to environmental protection more narrowly) and less capable of managing diffuse sources of pollution. Secondly, this growing dissatisfaction was linked to wider shifts of economic paradigms and political ideologies, increasingly embracing neoliberal ideas and the ‘regulatory reform’ agenda. Thirdly, periods of economic recession had increased the pressure to reduce costly regulation and look for alternatives. Fourthly, new environmental taxes and auctioning of pollution permits provided governments a sometimes badly needed new source of revenue. Fifthly, the EU has played a policy transfer role by promoting various NEPIs in EU-wide initiatives. Finally, as mentioned above, it has been claimed that the societal shift from government to 'governance' has led to an intensified use of forms of self-regulation and VAs.

Despite the strong push for greater use of NEPIs over the last couple of decades, there has also been strong opposition from some actors against certain types of instruments. Environmental non-governmental organisations (NGOs) and green parties were initially sceptical towards pollution charges and tradeable permits as they were seen as unethical in the sense of 'selling the right to pollute' (Majone 1976, p. 143) and VAs were seen as increasing the risk of capture by industrial interests (OECD 1999). More recently, however, at least economic instruments have become more accepted and actively supported, particularly their ability to apply the 'polluter-pays principle' in a more visible way (Golub 1998b; Tindale and Hewett 1998; Cook 2002; Jordan, Wurzel et al. 2003a). Industry representatives, on the other hand, have generally been supportive of the idea of more cost-efficient economic instruments on the whole, but have also exhibited some ambivalence and contradiction in the implementation of such instruments due to competitiveness and distributional concerns (Pearson 1995; Golub 1998b). As for political parties, Jordan, Wurzel et al. (2003a) find it surprising that centre-left parties have generally been more supportive of economic instruments than free market/centre-right parties, since these EPIs build on liberal economic ideas associated with latter types of parties. However, the introduction of new EPIs, regardless of type, should also be seen in the light of additional government intervention and general expansion of the public sphere, something which centre-right parties generally resist. Bureaucrats and agency officials, finally, have resisted widespread introduction of NEPIs to some extent, based on their professional backgrounds in traditional regulation and 'cultural antipathy' (Stavins 1998; Jordan, Wurzel et al. 2003b).

To sum up the evolving pattern of the use of EPIs, many analysts argue that the increased use of economic instruments promoted in the second wave was characterised by unrealistic expectations of how well they would perform and change the nature of environmental policy. Now, it is argued, this initial 'euphoria' has been replaced by more realistic expectations, a practical approach to solving implementation problems, and a good overall understanding of the relative strengths and weaknesses of different types of EPIs (see e.g. Huppes 1988, Majone 1989, Tietenberg 1990, Hockenstein, Stavins et al. 1997). This development has to some extent been mirrored in the case of VAs (see OECD 2003e). Increasingly, VAs are seen as complements rather than alternatives to traditional regulation, in order to ensure that they do not lead to regulatory capture or merely lead to changes that would have been made anyway under a business-as-usual scenario. It has also been observed that regulation continues to be the favoured and 'default' choice of instrument (de Bruijn and Hufen 1998 p. 20, Better Regulation Task Force 2000, OECD 2001c).

Arguably, the ideological debates over certain types of EPIs, such as economic instruments and VAs, have settled. Rittberger and Richardson (2003, p. 580) thus contend that "the 'mood change' has been rather ubiquitous and encompasses something of a rainbow coalition of interests". For example, in relation to economic instruments, Tietenberg (1990, p. 391) argues that "the issue is no longer *whether* they have a role to play, but rather *what kind* of role they should play". As described in chapter 1, governments have expressed an increased acceptance of, as well as proactive support for, a wide portfolio of potential instruments to choose from, on a case-by-case basis.

2.5 An increased interest in the mix of instruments

The fact that a single optimum approach or a 'panacea' instrument is unlikely to exist – neither in general nor in a particular situation – is one reason for using a mix of EPIs to address a policy problem (Gunningham and Gabrosky 1998). From a similar point of view, it has been observed that a combination of instruments is needed to address the oftentimes multiple policy objectives related to an environmental problem (see OECD 2001b p. 291, Naturvårdsverket 1997a, Helm 2003). Although the idea to construct mixes of complementary and mutually reinforcing types of EPIs to enhance overall effectiveness and efficiency, rather than rely on a single instrument, is not new (Rist 1998, p. 153), there is a burgeoning interest in both the design and study of EPI mixes (Golub 1998b p. 23, Barde 2002, Foxon, Makuch et al. 2004, OECD 2003d, EEA 2005).

From a practitioner perspective, the OECD has pointed to the complexity of many environmental problems and multiple policy objectives as reasons for focusing more strongly on mixes:

> "[b]ecause of the complexity of many of the most urgent pressures on the environment, their often inter-connected nature, and the limited understanding of some of their causes and effects, single policy instruments will seldom be sufficient to effectively resolve these problems. Instead, combinations of policy instruments will be required which target the range of actors affecting the environment, draw on synergies for realising the different environmental policy objectives and avoid policy conflicts, and which address any social or competitiveness concerns about the policy instruments" (OECD 2001b, p. 291).

The OECD initiative has resulted in both a 'policy packaging approach', which entails not just coordination and use of multiple instruments but also broad consultation and effective implementation and enforcement mechanisms (see OECD EAP Task Force 2003), and a work programme for systematically analysing instrument synergies (see e.g. OECD 2001b, OECD 2003d). According to Jordan, Wurzel et al. (2003a, p. 205) the UK and Netherlands have come furthest in Europe with experimenting with such mixes of EPIs. In particular, the emergence of climate change as a policy problem has highlighted the need to tackle complex problems in a comprehensive and diversified way (*ibid*.).

Second, from an academic perspective, the need to better understand how mixes are constructed and how instruments interact has also been recognised (Golub 1998b p. 23, Sorrell and Sijm 2003). It has been argued that 'mono-instrumental' study approaches are simply untenable, due to the interlinkages and complementarities of instruments in reality (de Bruijn and Hufen 1998, p. 24). A more critical stream in this literature has also discussed how policy mixes may in practice end up more as cases of 'policy congestion' (Majone 1989) or 'policy messes' (Sorrell and Sijm 2003).

Rather than relying on or heavily promoting one type of EPI, there is thus now a more concerted effort among practitioners in designing policy mixes to achieve synergy effects between instruments and to avoid conflicting or redundant instruments, with the ultimate goal of enhancing overall policy effectiveness (OECD 2001b). Johnstone (2004) identify several motives in line with this rationale: adapting to spatial or temporal heterogeneity of environmental impacts; reducing cost uncertainty; encouraging higher levels of compliance; technological market barriers or failures; extending regulatory reach; and addressing competitiveness and other concerns.

Rather few empirical analyses of policy instrument mixes have been made so far, especially with a systematic analytical framework. The OECD has provided some illustrations from the transport field and how to control motor vehicle emissions (OECD

2001b). In their study of policy mixes for 'sustainable innovation policy', Foxon, Makuch et al. (2004, pp. 14-15) quote the consultation on the revised EU Batteries Directive (91/157/EEC) as an example of a process where a mix is being contemplated. They note that besides from legislative targets, the revised Directive could also introduce a system of VAs and mandate member states to use instruments such as deposit-refund systems and charging systems. So far, Sorrell and Sijm (2003) analysis of CO2 emissions trading schemes in combination with carbon/energy taxes, support mechanisms for renewable electricity and policies to promote energy efficiency, is one of the more systematic studies. An analytical framework is used in which instruments are compared with reference to their scope, objectives, operation, implementation, and timing. This then allows for classifying the policy interaction of the trading scheme and various other instruments in terms of how target groups are covered: direct, indirect, or trading interaction. Using UK climate policy as example, it is found that there are several instances of overlaps, double regulation and double counting. Sorrell and Sijm conclude that the introduction of a carbon emissions trading scheme provide an opportunity to rationalise the policy mix but this may not always be undertaken, with the result that "a policy mix may easily become a policy mess" (p. 434).

To sum up, the policy mix literature so far has tended to have a *design* focus, aiming to establish how synergy effects can be achieved and how to avoid policy over-crowding, conflicts and unanticipated consequences. According to the OECD, "devising the right policy mix is a matter of careful analysis, design, fine-tuning and adaptation as well as coherence between different ministries (such as finance, agriculture and environment) and levels of government" (OECD 2001b, p. 133). In other words, how to achieve a good policy mix has mainly been seen as a technical design problem rather than a political question.

2.6 Understanding the mix pattern over time

In addition to studying instrument mixes as a design issue, there is also a need to understand how a *de facto* mix accumulates and evolves over time. From a political perspective, there is a need to examine what the mix's evolution means in terms of the relationship between the government and the target actors. One question for such a mix study is the *diversity* in terms of EPI types, and whether governments have delivered on their commitments to widen the range of instruments used. For this purpose, an EPI categorisation scheme was identified above (OECD 2001c). It is perhaps more interesting, but also more challenging in analytical terms, to analyse the *coerciveness* aspect of the instrument mix pattern. As described above, this is commonly seen as one of the most central features of an instrument, due to its implications for the continuous relationship between target groups and the government (see e.g. Gunningham and Gabrosky 1998; Macdonald 2001). To what extent can government intervene in private behaviour, while maintaining political support? Below, two different hypotheses are presented on how different instrument types represent different degrees of coerciveness and, consequently, why certain instruments are added to a mix in a certain sequence.

As mentioned above, Doern and Wilson (1974) have argued that the defining feature of alternative policy instruments is the 'degree of legitimate coercion' and that in liberal democratic societies governments would inherently prefer to use the least coercive instruments available. Thus, in the instrument typology elaborated by Doern and Phidd (1983) and reproduced in **Table 5** (see above), one would expect a movement from minimum to maximum degree of legitimate coercion (from left to right in the table). This is explained both by the smaller redistributive effects and greater constitutional ease associated with less coercive instruments, as well as the 'democratic sense' they make. Policy-makers would 'move up the scale' only as forced by recalcitrant target groups and/or social pressure to utilise more coercive instruments. This theory thus proposes that *in countries with more liberal governments (as opposed to interventionist) we would expect policy-makers to try to maintain instruments in the mix at a low level of coercion*. In a similar vein, and from a more overtly normative perspective, Gunningham and Gabrosky (1998) have proposed that the principle of least intervention should guide environmental regulation, and that governments should escalate up the 'instrument pyramid' only as necessary. Considering this hypothesis in the empirical context of this study we would thus expect that *in the municipal waste policy area, various informative measures and voluntary approaches would be introduced first, followed by grants and subsidies, and taxes and regulation as 'last-resort' instruments.*

This hypothesis has been considerably criticised. In addition to methodological problems (which will be discussed in chapter 4), it has been argued that the opposite pattern can often be seen, in that "governments frequently use coercive instruments in the first instance to convey the appearance of toughness to the population" (Howlett and Ramesh 1993 p. 8). Possibly, the environmental policy area is one where the public accepts, or even urges, initially forceful regulation, compared to other policy areas. Furthermore, the ideological assumptions have been criticised. Bemelmans-Videc and Vedung (1998) argue that rather than being deduced from a liberal political philosophy, the hypothesised pattern could be explained by the government's need to initially promote legitimacy of an intervention, with a view to gradually weaken the resistance of target groups to more coercive instruments. Despite these critiques, this hypothesis nevertheless offers a simple and clear benchmark pattern, that will be discussed in relation to Swedish and English municipal waste policy in chapter 6.

The second hypothesis is the more recent 'give-and-take' strategy proposed by van der Doelen (1998). He argues that the traditional one-dimensional notion of coercion as gradually increasing over the three classic instrument types, or 'control models' – from communicative control, through economic control, to judicial control – must be refined, as described above (see **Table 7** above). Policy instruments have two purposes, to legitimate a policy and to effectuate it. Usually, individual instruments perform better on one of these; "*stimulative* policy instruments (information, subsidies, and contracts) legitimate a policy, whereas *repressive* policy instruments (propaganda, levies, and orders and prohibitions) effectuate it" (p. 134, my itallics). Thus, to achieve higher overall effectiveness and efficiency of policy, *governments should 'give-and-take' among the stimulative and repressive variants of each control model*.

This hypothesis has not been subject to the same amount of critique as Doern and Wilson's, but a couple of potential problems can be identified. First, the possibilities for a 'take-give-give' strategy (i.e. after a certain threshold effect is achieved, target actors need only stimulative instruments to respond) or a 'give-take-take' strategy (i.e. once legitimated, target actors accept increasingly repressive instruments, as suggested by Bemelmans-Videc and Vedung above) being used are not explored. Second, a weakness with van der Doelen's hypothesis is that he does not discuss or predict giving and taking across the three control models, i.e. how stimulative information can increase the acceptance and effect of a repressive levy, or how a repressive order (e.g. emission standard) can be combined with a stimulative grant (e.g. technology development).

Thus, in addition to examining the *diversity* of the municipal waste policy instrument mix in terms of EPI types adopted in 1995-2005, the two competing propositions regarding the level of *coercion* by Doern and Wilson and van der Doelen will be analysed in chapter 6. While there are methodological problems with applying them empirically, such a study is still worthwhile in order to understand the political implications of broadening the policy instrument mix. The perspectives on policy mixes as mainly technical design problems need to be complemented with an appreciation of the long-term evolution of the instruments addressing an environmental problem and how the nature of government intervention may or may not have changed in political terms.

2.7 Conclusion

The purpose of this literature review was to provide a theoretical background for addressing the first research question of this thesis; *has the EPI mix become more diverse*? It was described how the use of EPIs has evolved from command-and-control regulation as a default tool, to the introduction of economic instruments, voluntary agreements and information measures, and finally to the promotion of comprehensive policy mixes. It was indicated that the pattern of EPI use is uneven across countries, but there are also difficulties involved in representing and measuring the significance of different EPIs in a meaningful way. This thesis aims at providing a better picture of the use of various EPIs in Sweden and the UK, both at an overall level in environmental policy and of municipal waste policy more specifically (chapter 6).

On a conceptual note, it has been suggested that the theory and practice of policy mixes is under-researched in the environmental policy field (Sorrell and Sijm 2003). The existing literature is to a large extent concerned with understanding instrument interactions, with the goal to design effective and efficient policy packages. The topic of policy mix seems to have been less studied from a political point of view, in terms of how the totality of instruments in a particular policy field creates certain relationships and resource exchanges among actors, and how actors debate and bargain over a new potential instrument in the context of an existing policy mix rather than from scratch.

The review of EPI definitions and typologies provided some insights into how such a more politically oriented analysis can be performed, and these will be included in the analytical framework for policy mixes developed in this study (chapter 4). Rather than developing an analytical framework for studying instrument interaction, the composition of different EPI types over time will be analysed in order to establish whether a pattern of EPI diversification emerges in line with the general trends described above. As mentioned above, one expected pattern would simply be an increased diversity of EPIs as a reflection of less biased and more comprehensive consideration of alternative EPIs. However, two other possible patterns will also be discussed in the empirical analysis. First, it will be analysed whether there is a trend of first choosing the least coercive EPI available and then gradually adopting more coercive EPIs (Doern and Phidd 1983). Second, it will be analysed whether the government instead have employed a ‘give-and-take’ strategy, by first choosing a stimulative instrument to legitimate a policy and then a repressive instrument to effectuate it (van der Doelen 1998).

To conclude, the research question relating to the ‘mix’ part of the thesis is: *has the EPI mix become more diverse?* Have more so-called ‘new’ EPIs been adopted, and, if so, which types, when and for what? And what are the patterns regarding the level of coerciveness of instruments added to a mix; have EPIs become increasingly ‘harder’ or ‘softer’? Furthermore, is a diverse mix an end in itself, regardless of its overall effectiveness? Finally, how do the patterns in the field of municipal waste policy resonate with the wider national EPI mix? Before the methodology used to investigate these research questions is described in chapter 4, the next chapter will explore theories relating to the process-oriented research question of this thesis, namely what determines the choice of EPIs.

Chapter 3

Understanding the Instrument Choice Process: Procedural Rationality in a Political Context

3.1 Introduction

In chapter 1, it was argued that EPI choice may be undergoing a depoliticisation process. It was described how many national governments have, at least rhetorically, committed to increasing the procedural rationality of the instrument choice process, as a way of ensuring that new and different forms of EPIs are considered as alternatives to more traditional instruments. This led to the formulation of the second set of research questions: *What determines the instrument choice process? In particular, to what extent does it approximate the procedural rationality ideal?* This chapter will review the political science and policy analysis literature on this topic and discuss the extent to which procedural rationality is a realistic ideal for the instrument choice process. Departing from the classical rationality model of instrument choice, the main critiques are summarised. However, it will then be suggested that broad-brush rejection may be misplaced. Trends in recent years of introducing more systematic RIA procedures and a culture of 'professional' policymaking may have had an impact beyond the purely rhetorical level. It will be argued that we need a more nuanced understanding of the extent to which an instrument choice process can indeed be 'rational'.

Apart from unpacking procedural rationality, alternative (or complementary) explanations of EPI choice are needed. The chapter therefore moves on to review the literature focusing on how various political and institutional factors shape EPI choice, both in terms of the process of choosing and the instrument choice outcome. A framework is suggested for organising such perspectives and theories based on their orientation towards either a predominantly structural explanation (macro-level) or a predominantly agent-centred explanation (micro-level), or somewhere in between (meso-level). The chapter is concluded by a summary of the main insights in the literature on the instrument choice process that will be used in the analytical framework for the empirical study.

3.2 Meanings of rationality and instrument choice

Using the word ‘rational’ in relation to EPI choice is contentious and there is a need to more precisely define the meaning of rationality. According to Bemelmans-Videc and Vedung (1998, p. 268), the rationalistic conceptions found in the classical administrative literature view decisions on policy instruments as “choices of means in goal-directed problem-solving processes” and as “neutral, even scientific enterprise[s]” (see also Howlett and Ramesh 1995). As described in chapter 1, some practitioner-oriented literature on EPIs discusses instrument choice in this way as a rather depoliticised ‘design’ issue determined by technical performance criteria or constraints, such as physical feasibility to change environmental processes, technical capacity to adjust production processes, and financial means to implement the instrument (see e.g. OECD 2001c p. 135; Naturvårdsverket 1999b; Richards 2000; Huppes and Simonis 2000; Yachnin, Gagnon et al. 2000; Sterner 2003). The implication is that a particular instrument is seen as the most rational in a particular setting, i.e. there is *a particular rational solution* to a problem. Clearly, such a perspective is not consistent with procedural rationality as described in chapter 1, which refers to *how a solution should be arrived at*. Arguably, these two views represent two different meanings of rationality; procedural and substantive (cf. Parsons 1995 p. 271, Rydin 2003 p. 78).

Substantive rationality thus refers to the situation when a particular policy instrument is the most optimal solution to a problem, based on a set of ranked choice criteria[16]. It is the instrument that is rational, rather than the process by which it is chosen, primarily. Typical criteria in the EPI context include environmental effectiveness, economic efficiency, equitability, legitimacy, and administrative and technical feasibility (see

[16] An example is the model provided by Richards (2000, p. 228). Assuming that cost-minimisation is the main policy objective, the decision criteria are structured as a ‘constrained optimisation problem’:

Minimise (PC + IC + TX)
Subject to the constraints:
Pollution abatement requirement
Legal constraints
Political constraints

where PC is production costs, IC is implementation costs and TX are public finance impacts.

Majone 1976, Jacobs 1991, Bemelmans-Videc 1998, Adger, Brown et al. 2003, Mickwitz 2003). In the welfare economics literature specifically, economic efficiency is the only one or highest ranked criterion. By implicitly or explicitly defining a criterion in this way, economic instruments often come out as the most rational instruments in comparison with others. Welfare economics could thus be seen as one such substantive rationality. Another such substantive rationality could be that assumed in rational choice theory, which stipulate electoral support or bureau-maximisation as the main (unofficial) choice criterion. Within the discursive approach to environmental policy, Rydin (2003) and Dryzek (1997) have identified other substantive 'rationalities', which are associated with certain 'default' instruments and choice criteria.

However, the quest for new EPIs by policy-makers outlined in chapter 1 has not stated that NEPIs are necessarily always the most rational instruments, in substantive terms. Instead, the rationality sought often relates to the procedure of decision-making. *Procedural rationality* thus refers to the process by which instruments should be selected and designed. Rather than specifying a certain criterion, the aim of procedural rationality is to ensure that *any* criteria are set and that they are clear. Procedural rationality stipulates an ideal decision-making procedure with a logical sequence of steps, namely:

- definition of the problem,
- setting objectives and choice criteria,
- identification of alternative instruments,
- impact assessment of those alternative instruments in relation to objectives and criteria,
- making a decision (see Howlett and Ramesh 1995 p. 140; Rydin 2003).

Policy-makers are seen as rational actors in control of the process and capable of applying measures to ensure procedural rationality in actual decision-making situations (Rydin 2003, p. 80).

In this thesis, I will focus on this second meaning of rationality, i.e. to evaluate the extent to which procedural rationality characterises the instrument choice process. In the context of EPIs, the underlying assumption here is that ensuring procedural rationality in an EPI choice process increases the opportunities for new EPIs being adopted, through widening and deepening the consideration of alternatives. The procedural rationality is viewed as an ideal process of instrument choice that policy-makers strive for.

3.3 Critique of procedural rationality

Unsurprisingly, the ideal of procedural rationality in the choice process has been strongly questioned in the policy instrument literature. It has been argued that the policy-making context is inherently imperfect or even chaotic. In addition to cognitive and resource constraints, the difficulties of ensuring procedural rationality – by some referred to as instrumental rationality – are augmented by the fact that it needs to take place in a political and social context rather than a purely analytical context. Rist (1998, p. 150) states that “what is faced… is the chaos of political agendas, confusion over objectives, contradictions in political messages, partial understandings of the societal conditions, self-aggrandizement among the political actors, cross-currents of desired economic and political outcomes, organised groups of stakeholders, and more or less capable and committed public-sector bureaucratic systems” (see also Howlett and Ramesh 1993 p. 13; Shapiro and Glicksman 2000).

Looking more closely at the critique, much of it stems from and refers to the long-standing debate in the public policy literature on the general rationality of decision-making (see e.g. Parsons 1995 p. 271ff; Dunn 2004 pp. 47-55). Classical contributions from scholars such as Simon (‘bounded rationality’), Lindblom (‘successive limited comparison’), Dror (‘normative optimalism’), and Etzioni (‘mixed scanning’) could indeed shed light on the phenomenon of EPI choice. However, the purpose of this chapter is not to review this substantive body of work, but instead focus on the more specific critiques of the procedural rationality as an ideal for the EPI choice process.

First, in relation to the *logical sequence of steps* stipulated within procedural rationality, it has been questioned whether ‘ends’ (policy objectives) can really be separated from ‘means’ (policy instruments), and whether the definition of ‘ends’ always precede the identification of ‘means’. Majone (1989, p. 115) argues that this ends-means separation is often false since “[i] the choice of means helps to alter the criteria by which the correctness of the means must be judged… [ii] policy goals are often defined in terms of the available means… [iii] when goals are vague or ambiguous and outcomes difficult to measure, the instruments used acquire a significance that goes well beyond their purely instrumental value". The close intertwining between selection of objective and instrument is also a key feature of Lindblom's (1959, p. 81) classical decision-making model of 'successive limited comparison' (as opposed to 'rational comprehensive'). Bemelmans-Videc and Vedung (1998, p. 269) elaborate on this point and claim that policy objectives are often only deliberately symbolic, hence more easily agreed upon, while the choice of instrument is where the ideological and political controversies are played out. For example, the waste hierarchy (see **Figure 1**, chapter 1) could be such a symbolic objective. Rather than a logical and linear process, it has thus been argued that the boundaries between stages of setting objectives and choosing instruments are likely to be blurred, they may run in parallel, and the process of instrument choice is iterative.

Second, the existence of *multiple objectives and criteria* in practice and the problems with lack of clear prioritisation have been recognised among practitioners (e.g. Naturvårdsverket 1997b). Naturally, the potential for simple and clear ends-means rationality decreases further when the objectives are conflicting and compromises are necessary. A kind of single-criterion technical evaluative exercise, such as that modelled in some welfare economic accounts with the economic efficiency criterion (see e.g. Richards 2000; Driesen 2003; Sterner 2003), is simply unrealistic according to several commentators (Elmore 1987; Majone 1976; de Bruijn and Hufen 1998; Daugbjerg 1999 p. 29; Vedung 1998 p. 39). In reality, several environmental and economic objectives may be relevant and conflicting criteria need to be balanced.

Third, the *identification and consideration of alternative instruments* may be narrow in practice, and far from the consideration of the 'myriad techniques' (theoretically) available to policy-makers. As quoted in chapter 1, Linder and Peters (1998, p. 36) state that 'systematic, comparative assessments' of alternative instruments are typically missing. Simon (1997, p. 126) has also pointed out the challenge of generating alternatives, as they are not simply given to the policy-maker. Various factors may also cause the alternatives that are identified to be only marginally different from each other (see Lindblom 1959).

There is thus a range of critiques pointing to the limited cognitive capacity of policy-makers, lack of resources and time, and the role of the political context for instrument choice. Various perspectives for understanding the role of political factors and how they may hinder or override procedural rationality will be reviewed later in this chapter. Before this review, however, a different kind of critique of the procedural rationality ideal also needs to be addressed, namely whether the appearance of procedural rationality in an instrument choice process is merely a *post-hoc rationalisation* of a decision taken under some other conditions. There may not be a genuine interest in organising a certain EPI choice process in a procedurally rational way, but it may simply be a means to ensure legitimacy of the final choice among policy actors.

The critique relating to *post-hoc* rationalisation has been developed in the discourse literature. Dryzek (1997, p. 70, 76) identifies ‘rationalistic policy analysis techniques’ to be part of an ‘administrative rationality’ discourse, which is based on the belief in the bureaucracy as the supreme form of social organisation. In a similar vein, Rydin (2003, p.76) identifies ‘procedural rationality’ as a discourse in environmental policy and planning and sets out to expose the self-image of the policy process as rational. The rationale of this discourse is traced to the need for bureaucracy to continuously re-legitimise itself and its authority, especially in postmodern times where the state is being ‘hollowed-out’ (see also Howlett 2000b). According to Rydin, the discourse of procedural rationality has several distinctive features: the assumed benefit of breaking up the policy process into steps and stages, the synoptic ideal of comprehensive data collection and analysis, and the ethos of the policy analyst as being in control (pp. 79-80). Furthermore, within the discourse of procedural rationality, “the messy interaction of interests within policy is ignored; methodology overcomes power” (p. 80). While substantive claims, such as increased economic efficiency or environmental effectiveness, also play in a role in legitimising policy, “the rationality of the policy process itself is seen as legitimating the activities of the public sector” (p. 78). The implication is that the steps associated with procedural rationality are not undertaken to *inform* the decision necessarily, but rather to *justify and legitimise* it. The key question is thus how genuine efforts to ensure procedural rationality really are.

3.4 A resurgence of procedural rationality?

In the light of this substantial critique of the plausibility of the procedural rationality ideal, the commitments in relation to the NEPI agenda for wider and more systematic consideration of alternative instruments seem unrealistic. Does this mean that procedural rationality is unattainable, and therefore uninteresting to examine empirically? Below, I will argue that some of the critique outlined above can be disputed. In addition, and as noted in chapter 1, some recent trends in practical policy-making regarding the use of evidence and formal RIA suggest that there may be a resurgence of procedural rationality.

The critique that efforts to introduce procedural rationality amount to nothing more than *post-hoc* rationalisation, and are part of a wider discourse serving to legitimise bureaucracy and public policy interventions as such, is difficult to refute. Ultimately, at the level of the individual policy-maker, it is impossible to prove if a certain action (such as the consideration of an alternative EPI) was taken to inform a choice or to rationalise a choice already (implicitly) made. However, it may be a misconception to dichotomise these two activities, i.e. informing *vs.* rationalising (c.f. Brunsson 2006; Parsons 1995 p. 440). Instead, there may be a pre-existing inclination towards a certain EPI, and procedural rationality measures both provide the final information needed and legitimisation of the decision. Perhaps these two functions should instead be seen as two ends of a spectrum. In order to determine where along such a spectrum a certain EPI choice process could be placed, one could study how the process was set up in advance. Were there plans to define the problem in detail? Were there plans to identify and consult stakeholders on a set of alternative instruments? Although such plans can be made with *post-hoc* rationalisation in mind, they arguably represent one step closer to the procedural rationality than a complete *ex post* construction of an, at best, coherent argument for the selection of a particular EPI.

Second, even if we accept that *post-hoc* rationalisation is the only purpose of introducing procedural rationality measures, or the only verifiable purpose, this still does not need to discredit the procedural rationality ideal for understanding policy-making. The alternative to a situation where there is some kind of *post-hoc* rationalisation of an EPI choice is chaos. A chaotic choice process implies that there are no discernible process steps, no clear logic or reasoning behind the choice, and no documentation of the deliberation (c.f. Jordan, Wurzel et al. 2003d). Clearly, a situation where a choice is at least rationalised significantly improves the scope for transparency and opportunities for questioning the logic behind the decision. Hence, *post-hoc* rationalisation represents one step closer to a situation of perfect procedural rationality, compared to a chaotic choice process.

What about the other critiques, which essentially point out the political and cognitive constraints preventing procedural rationality to characterise an EPI choice process? While they are convincing, the studies on which they build are primarily *descriptive* accounts of instrument choice. The procedural rationality ideal is a *normative* account of how instruments should be chosen. Importantly, these descriptive theories do not make it less interesting or relevant to again examine the extent to which a normative ideal has been achieved in practice. An interesting paradox, also, is that while the rational decision-making model is widely rejected, most policy-making guidance and also recommendations stemming from academic research still invoke the principles of procedural rationality (see Simon 1957 in Parsons 1995; Brunsson 2006). There seems to be a tension between recognising the chaotic and 'irrational' nature of the real world of policy-making, and still proposing measures that strive to achieve procedural rationality. For example, the guidance document 'Professional policy-making' by the Cabinet Office 1999) disqualifies the traditional stage model associated with procedural rationality, arguing that it is not realistic. Then a 'descriptive' model is proposed, which is even more inspired by elements of procedural rationality.

This suggests that the ideal type of procedural rationality has some attraction still in the world of policy and seems difficult to discard completely. Therefore, it is relevant to examine whether intentions to improve the EPI choice process have translated into practice. In chapter 1, it was also argued that recent empirical trends suggest that there might have been a step change in the extent to which procedural rationality is achieved. The question is, then, have these initiatives had any real impact on practice in concrete policy-making processes, beyond a rhetorical level? In chapter 4, these trends will be briefly described with the purpose to help unpack what procedural rationality means in operational terms and to construct an analytical framework for the case studies.

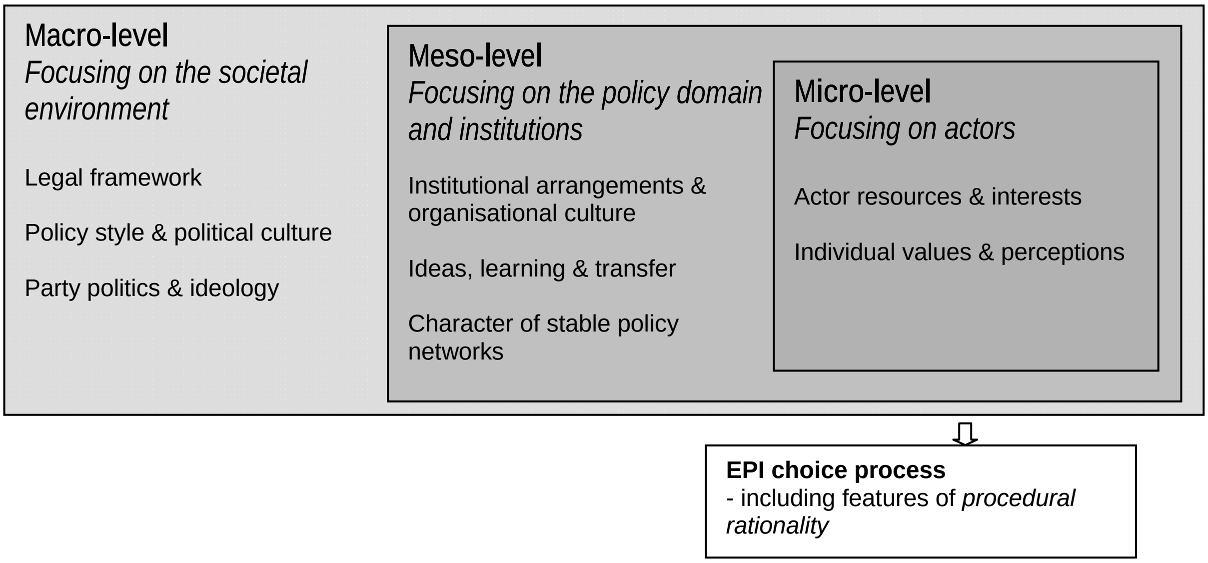
3.5 Theories on policy instrument choice

This thesis will thus assess the extent to which procedural rationality was achieved in the two case study EPI choice processes. However, other theories on instrument choice are needed to help explain the process and outcome. Procedural rationality takes place in a political and institutional context which needs to be understood. Furthermore, procedural rationality is 'neutral' in the sense that no specific objectives or criteria are dictated. Thus, other explanations of EPI choice need to be reviewed, in order to assess the extent to which various political and institutional factors hinder or facilitate the achievement of procedural rationality, as well as why and how certain policy objectives and criteria are prioritised over others. The rest of this chapter will present a literature review of alternative explanations of the EPI choice process[17] and thereby also a background for the analytical framework developed in the next chapter. The scope of this literature review is relatively wide, including both basic perspectives on instrument choice and more detailed theories in which specific predictions of instrument choices are made.

How can such a literature review of alternative variables and perspectives be structured? Both Linder and Peters (1989) and Daugbjerg and Svendsen (2001) use a *macro*, *meso* and *micro* level terminology to organise political and institutional factors explaining instrument choice (see also Peters 1998, p. 111ff). More specifically, Linder and Peters demonstrate how instrument choice can be analysed in terms of systemic (e.g. national policy style), organisational (e.g. organisational culture) and individual (e.g. cognitive factors) variables, corresponding to the macro, meso and micro levels. It is recognised that individually held perceptions and values "operate within a complex ecology of contexts, beginning with the decision-maker's immediate organisational circumstances and extending to features of their political system" (pp. 35-36). While recognising these contextual linkages, the authors do not attempt to trace the processes that link macro- and micro-level variables to find patterns of common variation or attribute relative causal significance. Instead, the 'Russian doll' metaphor can be useful to describe how the micro and meso levels are nested in the macro level. In **Figure 3** the most important political factors that have been identified in the literature and that will be discussed in the remainder of this chapter are listed in this way. Note that the micro-level refers to the level of individual policy-makers (Linder and Peters 1989), but also actor groups representing a common interest (such as an environmental NGO or a trade association).

[17] Note that this literature review is limited to theories and perspectives on positive choices of instruments and does not include the phenomenon of 'non-decisionmaking', i.e. decisions not to adopt a new instrument (see Bachrach and Baratz 1963; Schneider and Ingram 1990).

Figure 3. Factors influencing EPI choice



These levels reflect the structure-agency spectrum, i.e. whether instrument choice should primarily be understood as an outcome of wider political structures and societal preferences or as the result of the behaviour and preferences of individual policy actors. The level of analysis also reflects considerations made in relation to the scope of the policy instrument mix and the time period studied. Clearly, to explain the evolution of a broader pattern of instrument choice over time, macro-level theories are a more natural choice. It should be noted that using the macro-, meso- and micro-level terminology does not mean that focusing on one level excludes consideration of factors operating at other levels. Rather, the terminology helps to illustrate how emphasis can be placed at different places along a spectrum of levels of analysis. In fact, many of the studies reviewed below are comprehensive and consider variables at several levels, in a complementary way (see e.g. Kirschen, Bernard et al. 1964; Hood 1986, Böcher and Töller 2003; Holm Pedersen 2003). Some accounts consider in an explicit way the interaction and dualism of structural and agency factors, such as change-inhibiting national institutions and agents' advocacy of new EPI ideas (Tews, Busch et al. 2003; Jordan, Wurzel et al. 2003c).

3.6 Macro-level perspectives and factors

A common assumption in the literature focusing on the macro-level is that "policymakers operate in an environment in which they often have little to no influence in shaping the context that limits their policy options" (Rist 1998, p. 152). In the mix of perspectives and concepts reviewed below, the causality attributed to macro-level variables differs considerably and some accounts do not attempt to establish causal linkages at all, but merely emphasise the role of the broader context of EPI choice. Below, three key macro-level factors for explaining EPI choice in the literature are reviewed; the role of the legal framework, policy style and political culture, and party ideology and public opinion.

3.6.1 Legal framework

While legal regulation is an EPI itself, the overarching *legal framework* may in the short term preclude some instrument types or particular designs thereof (e.g. scope of the target group). Legal constraints at the *national level* can serve to protect citizens and firms from excessive state intrusion in behaviour. For example, Richards (2000) has analysed the US regulatory takings law, which defines under which circumstances a regulation counts as taking of private property and thus requires compensation for the loss. He states that US court rulings have recently come to find more circumstances as involving such a taking, while there are relatively few constitutional limits to the government's use of the taxing power. Another type of effect upon EPI choice is how the distribution of legislative mandate at different national constitutional levels (federal, state and local level) can limit the EPIs *de facto* available. For example, in the US, the federal government cannot directly regulate land use but have indirectly done so by requiring all states to develop and implement state-designed land use plans (Richards 2000). Also in Germany, the federal structure has put constraints upon which EPIs are feasible at the national level (Böcher and Töller 2003). While both Sweden and the UK are non-federal states, the central-local government distribution of legislative mandate plays a role in EPI choice, though.

International trade rules can also pose legal constraints to EPIs, such as when labelling and information disclosure EPIs are seen as trade barriers (see Appleton 1997; Beierle 2003; Sterner 2003 p. 206; Joshi 2004). Within the EU, *internal market legislation* promoting fair competition can limit the use of some economic instruments benefiting some groups or industries more than others. The use of subsidies and tax rebates are regulated under the State Aid rules in Article 87 of the Treaty of the European Union.

Another aspect of the EU legal framework is of course how much flexibility is allowed for member states to choose and design instruments to implement policies and legislation formulated at the EU level[18]. It has been predicted that the *subsidiarity principle* would have a strong effect upon EU environmental legislation (Golub 1996), *inter alia*, that it would lead to less precise legislation being adopted (i.e. more framework directives rather than regulations and decisions) and that it would be characterised by procedural requirements rather than substantive ones. The Commission itself recommended increased use of framework directives and common minimum standards in a 1996 report (Commission of the European Communities 1996a), as opposed to more detailed and binding legislation. There are divergent findings regarding the actual impact of the subsidiarity principle on recent EU legislation, though. It has been argued that increased scrutiny has led to a long and slow processes of preparing proposals (Shaw, Nadin et al. 2000). Jordan and Jeppesen (2000), on the other hand, argue that the subsidiarity principle has not led to a wholesale re-assessment of EU environmental policy, but rather the introduction of a technocratic set of procedures to encourage 'better law-making' (see also Philip 1998, p. 265). A quantitative survey by Jeppesen (2000) of the number of directives, decisions and regulations respectively that had been adopted in the environmental area each year in the period 1990-1998 confirmed that no trend towards less binding legislative instruments could be observed. Rittberger and Richardson (2003) have found that while NEPIs figure much in EU rhetoric (e.g. in EAPs), they are not strongly present in recent directives. However, their article does not discuss the division of legal mandate between EU and Member State level. Jordan, Wurzel et al. (2005c) find that national sovereignty has blocked wide uptake of NEPIs at the supranational EU level. So far, there is thus no one clear trend regarding the impact upon EPIs adopted at the supranational EU level and their character. However, the literature suggests that the national level is still relevant to consider for EPI choice, and it is here one finds the legal competence for higher uptake of NEPIs.

In summary, the national and international legal frameworks may prevent or strongly constrain the choice of instruments, in particular economic instruments such as

[18] Note that this study is concerned with *the national level*, thus only links between EU regulation and the scope for EPI choice at the national level are discussed here. For studies focusing on the pattern of EPIs specified and discussed at the *EU level*, see Weale, Pridham et al. (2000, p. 461), Philip (1998), Rittberger and Richardson (2003), and Jordan, Wurzel et al. (2005c).

environmental taxes and subsidies. Whether increasing EU environmental legislation has led to increased use of NEPIs and/or softer instruments at the national level seems to be unclear still.

3.6.2 Policy style and political culture

The concepts of policy style and political culture figure relatively frequently when the choice of policy instrument is discussed. Both notions are interpreted and used in diverse ways. Sometimes they are used interchangeably and sometimes the former is seen as a subset or component of the latter (see Peters 2001, p. 34). In its original conceptualisation by Richardson, Gustafsson et al. (1982), a national *policy style* was characterised in two dimensions: (i) whether an *anticipatory* or *reactive* approach is taken to problem-solving, and (ii) whether the government is inclined towards a *consensus* relationship with organised groups or an *imposition* relationship. For example, the UK is generally placed in the consensual/reactive quadrant (Parsons 1995, p. 187) and Sweden in the consensual/anticipatory quadrant (Ruin 1982; Lundqvist 1997). The core argument of Richardson et al. was that policy-making is not determined by the type of policy issue, as hypothesised by Lowi (1966), nor is it too sectorised for discerning an overall national style (see also Buller, Lowe et al. 1993 p. 180; Weale, Pridham et al. 2000 p. 141). The authors were interested in whether societies had ‘standard operating procedures’ for policy-making and if these procedures were associated with certain ‘legitimising norms’.

In the literature on public policy instrument choice, Howlett (1991) has recognised the role of policy style as a causal factor and argued that national policy styles are so pervasive that even scholarly work on typologies of instruments is influenced by the style in the nation states of the respective authors (in particular, Bruce Doern (Canada) and Christopher Hood (UK)). This leads him to state that “the relationship between policy styles and policy instruments must be elaborated more precisely” (p. 16). Peters (2000), meanwhile, has explored the relationship between policy instruments and public management style, which could be seen as a subset of policy style. Taking public management style as the independent variable, it is proposed that a ‘legalistic style’ is conducive to the adoption of command-and-control regulation while a ‘managerial orientation’ may be more conducive to instruments that “function through more complex interactions of social and political organisations" and also those that involve substantial discretion (p. 43).

In the environmental policy literature, the concept of policy style has been popular in comparative studies. Some characterisations of the British and Swedish policy styles have been summarised in **Table 9** as an illustration (for characterisations of the EU policy style see Mazey and Richardson 1993; Rittberger and Richardson 2003). However, it has not always been applied in line with the idea of 'standard operating procedures'. Some of the key policy or 'regulatory style' literature, for example Vogel (1986) and Arentsen (1998), focus on the implementation and enforcement rather than development of instruments, and is primarily concerned with the nature of pollution control command-and-control regulation rather than the broader repertoire of EPIs.

Table 9. Characterisations of the policy styles of Sweden and the UK

Sweden	United Kingdom
• Strong belief in the use of knowledge and secular rationality to shape the environment (Richardson, Gustafsson et al. 1982) • Managerial orientation rather than legalistic style (Peters 2000, p. 43) • Consensual, anticipatory, deliberative, rationalistic, open (Ruin 1982) • Cooperation, consensus, information, trust (Lundqvist 1997, pp. 48-49) • Corporatist (Bache and Olsson, 2001)	• Bureaucratic accommodation, bargaining, negotiation, consensus, reactive (Jordan and Richardson, 1982) • Continuity; gradual and reactive evolution; administrative informality and technical considerations rather than legislative rules and standards; pragmatic; incremental; tactical; fragmented; reactive; flexible; decentralised; negotiation; persuasion; accommodative; technocratic; voluntaristic; closed policy communities; (Carter and Lowe 1998) • Cooperative rather than adversarial; administrative discretion; closed and expert policy community; focus on policy process and machinery rather than policy content and planning targets (Weale, Pridham et al. 2000, pp. 180-183) • Informal, personalistic, pragmatic, empirical (Peters 2001) • Voluntarism, discretion, practicability. Reactive rather than anticipatory; tactical rather than strategic; pragmatic rather than ambitious; case-by-case rather than uniform; consultation and negotiation rather than imposition and confrontation (Jordan, Wurzel et al. 2003e)

In relation to environmental policy instruments, Andersen (1999) has proposed an analytical framework where policy style together with problem pressure are contextual variables, policy instruments and institutions the main independent variable, and ecological modernisation the outcome [19]. Andersen identifies three dimensions of 'policy style' (legal-constitutional system, administrative set-up and historical and

[19] This work is based on the argument that policy instruments are absent in Jänicke's framework for studying the capacity for ecological modernisation and need to be included.

cultural elements), and examines how variation in these have influenced the use of economic instruments for water pollution in four European countries. For example, the gaps created in an federal legal-constitutional system has led policy-makers to impose more stringent and inflexible instruments, such as in the US, Germany and the EU. It is also stated that more consultative policy styles has frequently resulted in the choice of more open framework regulations rather than stringent command-and-control regulation (p. 31). However, Andersen is somewhat ambiguous as to the exact role of policy style as a 'contextual variable', in causal terms. He explains that it "affect[s] the choice of pollution control strategies" (together with problem pressure) but later states that "it should only be regarded as a 'filter' for the implementation of environmental policy programmes" (p. 30). Thus, he also seems to subscribe to the view of policy style as a relevant concept primarily in implementation studies rather than policy formulation studies.

Like the idea of a national policy style, the concept of *political culture* also has diverse interpretations, often intimately (or indiscriminately) linked to factors such as policy style and political ideologies[20]. Linder and Peters (1989, p. 49) interpret political culture as whether a country has a 'statist' tradition or not. They propose that, *ceteris paribus*, countries with a more statist tradition, like Germany and Scandinavian countries, accept more intrusive policy instruments and centralised government intervention than less statist countries. However, it seems difficult to avoid circularity when making such an argument, in particular since the authors do not define what constitutes a statist tradition. Furthermore, the initial overview of recent trends in national EPI patterns in the previous chapter (section 2.4) suggests that there is an increased variety of instruments in both less and more statist countries.

Efforts have also been made to link political culture to 'ecological modernisation' as a paradigm or policy agenda[21]. Adoption of new EPIs (primarily economic and informative instruments as opposed to legal and administrative) has been seen as one of the key characteristics of ecological modernisation, since they are allegedly offering

[20] Peters (2001) defines 'political culture' as shared psychological orientations as to what constitutes good government in a society (p. 34). Referring to Pye (1965), he describes four dimensions of political culture: hierarchy and equality, liberty and coercion, loyalty and commitment, and trust and distrust.
[21] For in-depth discussions of ecological modernisation as a policy paradigm, see e.g. Weale (1992), Hajer (1995) and Mol (1996).

more dynamic incentives and implementing the principle of internalisation of environmental costs to a larger extent than command-and-control regulation (see Lundqvist 2000, p. 22). Different types of political cultures have been found to be differentially conducive to ecological modernisation as a policy paradigm, hence the potential of adopting new EPIs. According to Dryzek (1997 p. 151), a corporatist political culture with a consensual and interventionist policy style is the most conducive (see also Lundqvist 1997). European countries such as Germany, the Netherlands, Sweden and Norway are often placed in this category. The UK, on the other hand, was in the 1990s seen by scholars to follow a more traditional, pragmatic path (Weale 1992, Hajer 1995). Hence, one would expect more NEPIs in a corporatist country like Sweden compared to the UK. However, the observation that corporatist societies would be more conducive to ecological modernisation has also been challenged (Andersen 1999, Jahn 1999). Overall, though, theories on ecological modernisation and political culture have according to Andersen (1999) generally failed to incorporate the role of policy instruments.

To sum up, the notions of policy style and political culture are referred to in many instrument choice studies. It has been suggested that choices of instruments depend on the public management style (legalistic vs. managerial), the policy style (consultative vs. impositional), whether a ‘statist’ tradition exists, and whether a corporatist political culture has facilitated an ecological modernisation agenda. However, this literature has so far not been satisfactorily clear, coherent or direct in their explanation of choice of EPIs. First, the concept of policy style has been interpreted very widely and rather indiscriminately. Peters (2000) focuses on public management approaches, while Weale (1997, p. 93) states that it is difficult to detach from the dominant political ideology and strategy in the case of the neo-liberal British environmental policy from the early 1980s. In a later comparative European volume, Weale and colleagues interpret policy style as the structure of institutions and professional socialisation among policy elites (Weale, Pridham et al. 2000, p. 142), thus closely linking it to national institutional arrangements (see section 3.7.1 below). Also in Andersen's (1999) framework, it is difficult to distinguish policy style from institutional arrangements. Overall, Linder and Peters (1989, p. 16) see it as an unanswered empirical question whether “variations in policy style reflect traditions and institutional habits, or a more self-conscious analytical process”. Thus, while the original conception should not be discredited, the review above suggests that if applied it needs to be clearly defined in terms of concrete variables and separated from other potential causal factors (such as institutions or ideology) in order to be meaningful.

Secondly, policy style and political culture are not always clearly independent variables but seem to be used for describing an outcome, i.e. a certain pattern of adopted EPIs is described as a certain policy style. This causal circularity is apparent in Rittberger and Richardson (2003, p. 575), who analyse “the alleged shift in the European Union’s environmental ‘policy style’ – from a traditional regulatory style involving classic instruments of legal regulation, towards a new style based more on new kinds of policy instruments which are less impositional (Richardson 1982), more market-based, more reliant on cooperative decision-making, and which allow member states greater latitude in the implementation process”. This review thus suggests that policy style may be more usefully considered as a background factor (see e.g. Jordan, Wurzel et al. 2003e), than as a workable concept for explaining instrument choice.

3.6.3 Party politics and ideology

As explained in chapters 1 and 2, a basic assumption in this thesis is that inherent ideological biases towards certain EPIs have gradually eroded and policy actors are increasingly open to considering a wide range of EPIs. Nevertheless, some accounts of EPI choice based on party politics and ideology merit a review.

The only normative account on instrument choice found is that of Doern and Phidd (1983), which predicts that liberal democratic states have an innate preference for less coercive instruments, regardless of the political sympathies of the governing party (see section 2.6). From a descriptive perspective, one of Daugbjerg and Svendsen's (2001) explanatory models of the development of green taxation in Western Europe is that ‘*parties matter*’ (Daugbjerg 2001a)[22]. Daugbjerg’s basic argument is that green taxation is a socio-economic policy issue and that the traditional left-right dimension of party politics explain the adoption of this specific EPI. He hypothesises that left-of-centre parties tend to perceive green taxation as an acceptable policy instrument since it does

[22] The other two explanatory models employed in their study are ‘group mobilisation and rent seeking’ and ‘policy networks’.

not directly burden their core constituency, typically workers, and it is also a way of demonstrating environmental concern and raising state revenue (see also Holm Pedersen 2003). Right-of-centre parties, on the other hand, are generally affiliated with producer interests, meaning that they usually prefer softer instruments such as VAs. If their core constituencies are exempted from taxation, however, they may support a green tax. Centre parties tend to take intermediate positions. These hypotheses were tested through an analysis of the status of green taxation up until 1997 in five European countries with left and right governments, with the conclusion that they were valid. For example, Sweden with its long dominant left-of-centre government has relied more on green taxes and placed greater tax burdens on producers, than the UK, where green taxes have been very rare, and Denmark, where green taxes are common but target consumers rather than producers.

Considering the methodological rigour of Daugbjerg (2001a) study, the conclusions appear strong. However there are some difficulties with testing similar hypotheses in the context of this thesis. First, his study is limited to analysing taxation as an EPI. His underlying question is arguably whether parties wish to extend the overall tax base to environmentally-related activities, rather than whether parties will prefer a tax over other possible EPIs to address a given policy issue. His inquiry does not consider other types of instruments addressing the given policy issues (e.g. CO2 emissions, nitrification, water discharges). Second, the impact of green taxes on producer interests, as a basis for political parties' acceptability of taxes, is not problematised. International variation in the relative importance of energy-intensive and polluting manufacturing industry in the overall economy is not elaborated upon. Furthermore, it is not clear why left-of-centre parties with workers as their core constituency would support heavier taxation of industry, and thereby potentially put industrial employment at risk. For example, in Sweden the industrial trade associations and trade unions have successfully mobilised together to ensure significant exemptions and rebates in energy and CO2 taxation for industry (Svendsen 2001).

Jordan, Wurzel et al. (2003a) also bring attention to the role of political parties in introducing green taxes, labelled together with VAs and ecolabelling as 'new EPIs' (NEPIs). However, the results of the country case studies are somewhat contradictory. First, it is stated that neo-liberal ideas about deregulation have been important drivers of the increasing uptake of NEPIs in several European countries (p. 203). Then, it is concluded that the country studies show that centre-left parties “helped tip the balance towards NEPIs in a number of countries” (p. 205). It is thus unclear if centre-left parties have picked up neo-liberal ideas or if they are referring to two parallel trends. However, it is later specified that the promotion of NEPIs by centre-left parties is primarily related to taxes, rather than the other types of NEPIs, highlighting the need to not bundle these diverse instruments together.

Continuing their argument, Jordan, Wurzel et al. (2003a, p. 205) argue that it is “somewhat of a puzzle” that centre-left governments have been more enthusiastic of economic instruments than free market/centre-right governments that dominated politics in the 1980s and early 1990s. According to the authors, “the former have traditionally supported ‘big government’ and its corollary regulation, whereas the latter advocated the very liberal economic ideas that economists often draw upon to advocate MBIs [economic instruments]” (p. 205). Instead, they find that most free market/centre-right governments have promoted VAs as an EPI. However, this is arguably a false paradox, since two different questions are simultaneously addressed; (i) should the government intervene at all and, (ii) if so, by which instruments should it intervene? The preference for VAs over economic instruments of centre-right governments does not seem inconsistent with the centre-right ideal of minimal public intervention. Neither is it immediately clear why regulation is more of a corollary to ‘big government’ than redistributive measures such as some environmental taxes.

An important finding of Jordan, Wurzel et al. (2003a) is the role of *green parties*. It is found that the adoption of green taxes in Germany and the UK in the mid-to-late 1990s was associated with the rise of the so-called Third Way political philosophies. In Finland and France (as well as Germany), meanwhile, green parties are found to have played a role in championing green tax reforms.

Thus, it has been found that political parties and ideology matter for EPI choice. In particular, the choice of environmental taxes as an EPI has been analysed in this literature. Daugbjerg's (2001a) theory, which is consistent with Jordan et al’s (2003a) findings, seems to explain the adoption of green taxes, but it does not address how a more comprehensive policy mix is developed towards a certain policy issue or how parties would choose between alternative EPIs. Therefore, the basic assumption in this thesis that ideological biases for or against certain EPI types have gradually been eroded remains.

3.7 Meso-level perspectives and theories

The literature focusing on macro-level factors for explaining instrument choice provides important insights. For example, it was indicated that the legal framework and party-political setting can play important roles for the selection of EPIs. However, there are also some general weaknesses in this literature. On an empirical note, several studies have focused exclusively on environmental taxes as an EPI, and not considered a choice situation where other types of EPIs could potentially have been selected. On a theoretical note, some of this literature is imprecise regarding the driving forces of instrument choice, which results in circular arguments, i.e. that a policy style predisposes instrument choices and that the chosen instrument repertoire is then a constituent element of the particular policy style.

Compared with the macro-level studies reviewed above, some other strands of the instrument choice literature assume there is some room for agency, alongside structural driving forces, in shaping instrument choice processes. However, the driving forces in these *meso-level* frameworks and theories are not the individual actor's or organisation's interests or preferences. Instead, they refer to more or less stable features of an organisation's mode of policy-making, such as national institutions and organisational culture, and the responsiveness to new EPI ideas, or to the nature of a stable actor configuration, such as generic characteristics of the policy network in a given policy arena. Generally, these frameworks are more adapted to studies of a particular policy area (such as the environmental policy arena), rather than the broader pattern of instruments adopted in society.

3.7.1 Institutional arrangements and organisational culture

Several theorisations of public policy instrument choice have drawn upon new institutionalism and organisation theory. Bagchus (1998) refers to a set of historical and sociological institutionalist propositions regarding the policy process. He distinguishes this approach from the rationalistic 'instrumentalist perspective' by identifying the following shifts in focus (pp. 52-53):

- from present to past – choices made in the past restrict the availability of future options, due to historic exclusion (material or psychological) and 'sunk cost';
- from design to evolution – intentional behaviour is questioned, and instrument choice is seen as an incremental development only to a small degree controlled by the actors involved; and
- from result to process – actors are not guided by the effectiveness of instruments when choosing them, but ensuring a 'logic of appropriateness' (March and Olsen 1989) and fulfilling their expected roles in the process.

As an example of this approach, the comprehensive analytical framework proposed by Linder and Peters (1989) includes 'organisational culture[23]. They state that "not only does the collective memory of an organisation tend to be associated with the repetitive use of certain instruments, but the very nature of institutions may limit their choices" (p. 42). The *path dependency* in instrument choice is thus emphasised. It is argued that the time period when an organisation was formed and the set of values and symbols dominant at the time will persist and predispose the organisation towards favouring some instruments. For example, organisations formed during the expansion of the public sector may favour cash grants or direct provision, while organisations formed during periods of fiscal restraint may favour cheaper regulatory instruments.

Linder and Peters' (1989) list of causes of variation in choice of instruments also includes the *professional backgrounds* and patterns of socialisation of bureaucrats within the national institutions. They do not elaborate upon this, but the importance of professional background has also been noted by for example Jordan, Wurzel et al. (2003a, p. 206), who link the reliance on regulation to the legal and scientific background of civil servants in many European countries. Also Holm Pedersen (2003) found that professional background and the associated institutionalisation of the knowledge base mattered for the uptake of green taxation as an instrument.

[23]Linder and Peters (1989) use the terms organisational culture and institutions an interchangeable way.

Institutionalist accounts of EPI choice have thus emphasised factors such as historical path dependency, the importance of professional backgrounds of policy-makers, and the dominance of the 'logic of appropriateness'[24]. These factors are all embedded in the macro-level national policy style discussed above, but can also be specific to the environmental policy sector rather than characterise the whole government apparatus. In a couple of studies, national institutional arrangements in the environmental policy arena have also been considered as a *counteracting force to the uptake of new EPI ideas*[25]. Ideas as a factor in EPI choice will be considered more in-depth in section 3.7.2 below. Jordan, Wurzel et al. (2003b) argue that the repertoire of EPIs is actually part of national institutions and "the fact that these repertoires provide appropriate solutions to national problems gives them an institutional embeddedness that is hard to dislodge unless the instrument in question is manifestly dysfunctional" (p. 19). Similar to Bagchus' (1998) propositions above, instrument change is therefore likely to be slow, incremental and path dependent, since "actors invest substantial time and resources adapting to particular policies and tools" and "actor preferences are derived endogenously on the basis of what is appropriate" (pp. 19-20). In addition, there are increasing returns to be reaped from sticking with traditional regulation and avoiding bureaucratic costs involved with shifting to NEPIs, contributing to the 'powerful refraction' of external political pressure for change. Based on these general propositions from the institutionalist literature, Jordan et al. propose that institutions act as barriers to innovation; that instruments that work with the grain of national institutions are more likely to be adopted; and that the 'filtering effect' of national institutions upon NEPI ideas may only become fully apparent after studying instruments through into the implementation stage.

On the basis of their country case studies, Jordan, Wurzel et al. (2003a, p. 219) conclude that the institutional arrangements are indeed important for shaping EPI choice to a great extent, since there is significant international variation in the design of generic NEPI types. However, institutional theory fails to explain the fact that there has actually

[24] Böcher and Töller's (2003) study is an example of an understanding of institutions as more formal and explicit decision rules, namely those related to legislative powers in a federal system, Cabinet structure and veto points, and the influence of the EU legal framework.
[25] In their *Public Administration* article, Jordan, Wurzel et al. (2003d) contrast the 'ideas dominant' and 'settings dominant' theories with a third explanation; 'chaos dominant'. The latter theory stipulates that instruments look for policies and vice versa. The search process is *ad hoc* and involves a pluralistic mix of actors.

been a major shift to NEPIs. The authors explain this shift by the emergence of advocacy coalitions increasingly supportive of NEPIs (see below). Neither does the institutionalist theory account for ‘old-fashioned’ political factors such as weak green parties, the political power of the Treasury vis-à-vis the environment ministry, and industry suspicion. Hence, they suggest there is a ‘draw’ in the explanatory power of institutional theories vs. ideational theories.

The conclusions by Tews, Busch et al. (2003) are slightly different. Making a similar theoretical argument on the ‘filtering effect’ of national institutions, they found that three of the four types of EPIs included in the empirical study did diffuse relatively rapidly (i.e. ecolabels, national environmental policy plans or strategies for sustainable development, and free-access-of-information provisions). The explanation of the slower diffusion of the fourth type (energy/carbon taxes) emphasises the conflict potential of this policy instrument rather than obstacles in the national institutional arrangements.

The literature on the role of institutions in instrument choice is thus very much in agreement in that they predispose policy-makers towards certain types of instruments, due to path dependency, administrative investments made in specific types of instruments, bureaucrats adhering to the ‘logic of appropriateness’, and specific patterns in the professional backgrounds of bureaucrats. This agreement, coupled with some supporting empirical evidence, suggests that institutional arrangements and organisational culture are key factors to consider when explaining EPI choice. However, there are also some difficulties of the institutionalist perspective. First, the standard phrase in the literature is that institutions ‘limit the choice of alternative instruments’ (see also Hanf and Jansen 1998) and most authors see them as intervening variables, or filters, rather than truly independent variables and active driving forces upon instrument choice. Therefore, an exclusive focus on the role of institutions does not seem able to provide the full story on EPI choice. Indeed, the studies reviewed above found that EPI change took place *despite* national institutional arrangements favouring status quo.

Second, the institutionalist studies reviewed here do not unpack in a detailed way what is meant by an institution, especially not how it has and can be empirically operationalised. Should one examine an organisation’s formal procedures and routines, or try to capture the norms and informal practices of bureaucrats, or both? Like the policy styles concept (see section 3.6.2), institutional arrangements risk becoming a rather indiscriminate 'residual' variable.

Finally, adopting an institutional perspective entails an implicit assumption that the governmental institution is a rather unitary and autonomous actor in the instrument choice process, and in particular the bureaucratic machinery and its procedures, routines and norms. The role of interests and interaction with stakeholder groups is not addressed. In section 3.7.3 below, the policy network perspective, which disagrees with this assumption, will be reviewed.

3.7.2 Ideas, learning and transfer

As described above, the role of institutional arrangements for EPI choice has often been contrasted with the role of *ideas*, where the latter represents the driving force and the former the 'filter' for policy change. Below, some of these ideational theories will be briefly reviewed[26], including some studies within the policy learning and transfer literature. The common denominator in this literature is that it examines ideas fed to a particular policy arena over time (meso-level), as opposed to an idea relating to a specific solution in a given choice situation (micro-level).

In their comparative study of NEPIs, Jordan, Wurzel et al. (2003b) draw on Hall's (1993) social learning theory and Sabatier's (1998) advocacy coalition framework. It is argued that "the struggle between these coalitions provides the primary motor of policy change" (p. 19). Furthermore, "at any one time, there is likely to be a dominant coalition which sets the intellectual framework… within which policy decisions are made, and defines a series of minority coalitions" (*ibid*.). Advocacy coalitions were not studied in a systematic way across all country case studies, though. It is still concluded that general advocacy of NEPIs has broadened since the 1970s when only a few academics and think tanks promoted them (Jordan, Wurzel et al. 2003a, pp. 217-218). For example, environment ministries have become increasingly supportive. Otherwise, advocacy varies with different types of NEPIs. For example, environmental NGOs and trade unions have promoted environmental taxes but remained sceptical of VAs and tradable

[26] For a more extensive review, see Holm Pedersen (2003).

permits. Overall, though, the authors conclude that advocacy coalitions “do not explain why some instruments were selected while others rejected” (p. 218).

Similarly, Holm Pedersen (2003, pp. 27-28) concludes that the idea of externality taxation in itself has limited viability and that other factors have been central to the introduction of environmental taxation, namely opportunities for Social Democratic parties to appeal to a green electorate and for policy entrepreneurs in the administration seeking new sources of tax revenues. Also Daugbjerg and Pedersen (2004) examine how the idea of green taxation has been implemented differently in national settings. Specifically, they compare pesticide and CO2 taxation in Denmark, Sweden and Norway. Their main conclusion is that ideas “set some kind of policy agenda, but they do indeed leave considerable room for politics” (p. 245). Intervening variables, such as the nature of state-producer policy networks in their case, are more important for explaining variation in the uptake of green taxation.

The literature thus suggests that studying ideas alone as a driver of EPI choice in a particular setting is misguided. The role of ideas has also been examined from a more historical and evolutionary *policy learning* perspective. Howlett and Ramesh (1993) identify three dimensions of the learning process – “inter-temporal, inter-sectoral and cross-national” – that affect the choice of policy instrument. Regarding the inter-temporal dimension, the choice of instruments is seen as a cumulative process where past experience enables decision-makers to “form a rational and instinctive assessment of which instrument works best under what circumstances”, i.e. potentially reinforcing the national policy style and contributing to path dependency (see above). Somewhat contradictory, May (1992) proposes that there is an innate preference for new instruments; “the corollary for policy learning is presumed superiority of an untested policy instrument for addressing a given problem”. In any case, Howlett and Ramesh (1993) argue that in normal periods, decision-makers can be expected to fine-tune instruments, while in periods of paradigmatic change they can be expected to rethink their basic attitudes to various policy instruments.

The learning literature also identifies different types or levels of learning, ranging from instrumental learning – implying changes of instrument types, designs and or settings – to more fundamental learning – implying revised policy objectives and problem definitions[27]. In the context of this study, we are more interested in the former type, where EPI change thus would be an indicator of *instrumental learning* (see May 1992). For example, Fiorino (2001) discusses changes of EPIs as representing instrumental, or technical, learning. May (1992, p. 344) finds evidence of instrumental learning ("changes in preferred tools with stronger emphasis on recovery and recycling") in hazardous waste policy from 1976 to 1984, whereas no such evidence is found in the pesticides policy from 1972 to 1987 ("basic tools and implementation relationships unchanged"). Overall, though, this literature provides a framework for comparing instrument change with other forms of policy change, rather than explain *why* a certain instrument choice is made.

Change of EPIs over time has also been analysed from the perspective of *policy transfer* and diffusion, i.e. where ideas and experiences are exchanged internationally and induce adoption of new types and designs of EPIs at the national level[28]. Many authors refer to international ideas and trends and the role of international policy brokers such as the OECD as background factors (see e.g. Jordan, Wurzel et al. 2003a; Weale, Pridham et al. 2000). Tews, Busch et al. (2003) have tried to explain the 'rapid diffusion' of four types of new EPIs: ecolabels, energy or carbon taxes, national environmental policy plans or strategies for sustainable development, and free-access-of-information provisions. The authors conclude that "the adoption of environmental policy innovations is more likely if these policy innovations figure prominently on the global agenda" (p. 592). The speed of diffusion is related to the institutionalisation of transfer at the international level, through an international organisation, and to the conflict potential of the issue at the national level. Another important argument they put forward is that there may not only be an efficiency rationale to transfer, but also a legitimacy rationale, which works in two ways. First, in the face of uncertainty, national policy-makers may wish to rely on 'best-practice' instruments promoted internationally since

[27] May (1992) identifies three forms of learning: instrumental learning (viability of policy interventions or designs), social learning (social construction of a policy or problem), and political learning (strategy for advocating a given policy idea or problem). Hall (1993) defines three forms of policy change in the context of social learning: first-order change (adjustment of the settings of instruments), second-order change (altering the instruments), and third-order change (shift of overall policy goals). Glasbergen (1996, in Fiorino 2001) identifies three types of policy learning: technical learning (a search for new policy instruments in the context of fixed policy objectives), conceptual learning (a process of redefining policy goals and adjusting problem definitions), and social learning (focuses on interactions and communications among actors).

[28] For a review of the theoretical literature on policy transfer, see Dolowitz and Marsh (1996) and Evans and Davies (1999).

they may be perceived as more legitimate by the target actors (rather than necessarily a more rational or efficient solution). Second, legitimacy is also a driving force for countries to contribute to transfer of their own respective EPI innovations. In addition to 'regulatory competition', they suggest there is 'ideational competition' as a result of countries wishing to strengthen their image as legitimate members of an environmentally responsible global society.

Smith (2004), on the other hand, concludes that the role of policy transfer was limited in the adoption of three key climate policy instruments in the UK; the climate change levy, negotiated agreements on energy use, and the (then) voluntary scheme for carbon emissions trading. All these instruments had antecedents overseas. However, "interests quickly spilled into the climate policy arena and acted as constraints for the pure transfer of policy lessons" (p. 89). In particular, the election of New Labour in 1997 facilitated climate policy agenda-setting and business lobbies were influential in modifying the instruments to their benefit. Hence, Smith's findings suggest that there is need to consider also party-political factors and interaction in the policy networks when explaining the choice of new EPIs.

To sum up, how useful are ideas, learning and transfer approaches to the study of EPI choice and design? First, idea-based approaches seem to have been used in studies focusing on a specific type of EPI and variation in its adoption in different national contexts, rather than studies interested in EPI choice for a particular policy issue or area (such as municipal waste). While the latter approach would by no means be impossible, the empirical evidence so far suggests that ideas are less significant as a causal drivers than policy actor interests and political institutions (Daugbjerg and Pedersen 2004, Smith 2004), or subject to a significant 'filtering' effect at the national level (Tews, Busch et al. 2003, Jordan, Wurzel et al. 2003a). Second, these approaches, in particular the learning concepts, generally seem better suited to studies of instrument change patterns over time, than to studies exploring the mechanisms involved in a particular choice situation. Finally, the transfer studies showed that ideational competition can be a driving force on EPI choice, but in the end national politics play an important role. Overall, this review suggested that ideas as a single factor cannot explain EPI choice, but needs to be combined with institutional arrangements, party-politics, etc.

3.7.3 Character of stable policy networks

Both the institutional and ideational perspectives on EPI choice described above are rather state-centric. It is assumed that EPI choice is primarily a matter for the government apparatus rather than interest groups, whether the main mechanism is policy-makers following bureaucratic norms and routines or taking up new EPI ideas from experts. Several instrument choice theories disagree with this assumption and argue that the choice process consists of dynamic interaction and bargaining between actors, rather than autonomous design by politicians or bureaucrats. Most of these theories are concerned with the micro-level choice situation, i.e. when an individual instrument is chosen (see section 3.8). However, Bressers and O'Toole's (1998) theory is concerned with the character of the policy network in a particular policy arena over time, and its implications for instrument choice. Importantly, more or less constant and continuously reproduced properties of the network are in focus and the network is not simply used as a heuristic to describe the interaction between actors driven by their interests in relation to a particular issue. In this way, it is more appropriate to think of their theory as a meso-level explanation of instrument choice.

Bressers and O'Toole (1998) propose that two key features of the policy network determine the choice of policy instrument, or more specifically ‘institutional features’ of instruments. The two network characteristics are *cohesion* and *interconnectedness*. A range of hypotheses are then formulated in relation to six instrument characteristics: normative appeal, proportionality between target group behaviour and government reaction, provision/withdrawal of resources, freedom to opt for/against application, bilaterality/multilaterality, role of policy-makers in implementation (see **Table 10**). Cohesion is defined as “the extent to which individuals, groups, and organisations emphatise with each other’s objectives insofar as these are relevant to the policy field” (p. 219), and stems from shared values and worldviews. Interconnectedness is defined as “the intensity of actors’ interactions” (*ibid*.), and refer to the degree to which (semi-) private organisations take part in policy formation by means of formal consultative structures or by being members of committees or advisory bodies. The authors argue that interconnectedness can be seen as a ‘structural’ characteristic and cohesion as its ‘cultural’ counterpart, and there is allegedly no reason to assume that they necessarily co-vary. The ‘autopoiesis’ (self-creation and –organisation) of social systems is taken as a theoretical foundation, hence the central proposition is that "the more an instrument's characteristics help to maintain the existing features of the network, the more likely it is to be selected during the policy formation process" (p. 220).

Table 10. Bressers and O'Toole's hypotheses on instrument characteristics

	Strong interconnectedness of network	**Weak interconnectedness of network**
Strong cohesion of network	• Absence of normative appeal to target group • Proportionality • Net provision of additional resources to target group • Freedom for target groups to opt for or against application of instrument • Bilateral or multilateral arrangements • Implementation by policy-makers or closely affiliated organizations • *Example: US agriculture sector – subsidies and price support, research, targeted information and advice*	• Absence of normative appeal • Proportionality (weaker) • Provision of additional resources to target group • Considerable freedom for target groups to opt for or against application of instrument • Absence of bilateral arrangements • Implementation by policy-makers or closely affiliated organizations • *Example: Dutch economic policy – investment subsidies, research subsidies, information through intermediary organisation*
Weak cohesion of network	• Normative appeal • Proportionality • Limited withdrawal or resources • Absence of freedom for target group • Many bi- or multilateral arrangements • Implementation by policy-makers or affiliated organizations • *Example: Dutch environmental policy – agreements and covenants*	• Normative appeal • Absence of proportionality • Withdrawal of resources from the target group • Limited ability of target group to opt for or against application • Absence of bilateral arrangements • Implementation characterised by involvement of parties other than policy-makers and closely affiliated organizations • *Example: US environmental policy - regulations*

Source: Adapted from Bressers and O'Toole (1998, pp. 230-233).

Bressers and O'Toole illustrate the expected instrument choices with examples of each of the four kinds of policy networks and conclude that their hypotheses are valid. The predicted instrument features suggest that a strongly cohesive and interconnected policy network would result in the adoption of more voluntary instruments (e.g. environmental VAs) and less costly instruments (e.g. environmental subsidies). Conversely, a weakly interconnected and incohesive network would tend to adopt more costly and mandatory instruments, such as regulation or taxes. Implicitly, this suggests that more integrated policy networks benefit the target group in terms of instruments that are less costly to them or less coercive.

The propositions by Bressers and O'Toole constitute a major effort to complement and generalise from studies concerned with unique case histories and circumstances. The usefulness of this framework is demonstrated for instrument choice studies at the meso-level, i.e. when instrument choices are made repeatedly over time within a particular policy arena. However, the level of generalisation aspired to also involves some problems. First, the authors admit that the two independent variables, cohesion and interconnectedness, may be difficult to distinguish from each other. Furthermore, the key effect of each one respectively is not clarified. It appears that cohesion is important for the 'net provision of additional resources to the target group' (benefit or cost), as well as the need for 'normative appeal'. Interconnectedness, meanwhile, seems more important for the implementation arrangements and whether 'bilateral arrangements' can be made. It is possible that the model could be simplified and these key effects thereby highlighted. Second, in a similar vein, the instrument attributes are rather many and it is possible that they could be collapsed into fewer categories. For example, 'proportionality', the 'wihdrawal/provision of resources' and 'freedom to opt out', are all indicators of the expansion or limitation of freedom for target groups in how they behave. It could be argued that expected *types* of instruments should be included, but the authors deliberately avoid this, due to inconsistencies within an instrument type with reference to these attributes. This approach is thus in line with the need to focus on design attributes of EPIs, in addition to type, identified in chapter 2.

3.8 Micro-level perspectives and theories

The interests and interaction of policy actors in the course of an instrument choice process have been analysed more closely in some studies. In these approaches focusing on the micro-level choice situation, a larger degree of human agency is assumed to govern instrument choice, and individual or organisational interests and preferences are explored. The concept of a policy network has been used also at this level, but in these cases it refers to a discrete formation of actors around a particular issue and is used as a heuristic to described actor involvement rather than as a driving force itself (cf. Bressers and O'Toole 1998, section 3.7.3). Below, two different micro-level approaches are reviewed. Firstly, a couple of studies focus on the balance of power between regulator and regulatee, taking resources and interests as the main driving forces behind the behaviour of actors. Secondly, Linder and Peters (1989) have argued that there is a need to focus on the level of individuals' values and perceptions.

It should be noted that the literature review presented in this chapter is delimited to the political science and policy analysis literature on instrument choice, and EPI choice in particular. A range of micro-level theories on EPI choice have also been put forward within neoclassical economics and public choice. These theories refer both to the choice between different types of instruments and to the choice of stringency of an instrument, in particular the rates of green taxes. On the 'demand' side, the preferences and 'rent-seeking' behaviour of various interest groups have been modelled by e.g. Dewees (1983), Keohane, Revesz et al. (1997), Dijkstra (1998) and Svendsen (2001). A common finding is that command-and-control regulation is often preferred over economic instruments. On the 'supply' side, public choice studies have tried to understand the incentives of legislators. One proposition is that states choose instruments that provide concentrated benefits to marginal voters while spreading the costs to the entire population (Buchanan 1980, in Howlett and Ramesh 1993). Rather than assuming that electoral or bureau-maximising incentives determine the behaviour of policy-makers, Ciocirlan and Yandle (2003) discuss the revenue-generating incentives of government when choosing green taxes as EPIs. Perhaps the most comprehensive political economy analysis of EPI choice is that by Keohane, Revesz et al. (1997), who are concerned with both the 'demand' and 'supply' sides of EPIs. It is proposed that command-and-control regulation is the preferred choice for many interest groups, hence a very likely outcome. On the supply-side, legislators can be expected to prefer regulation since they can be supplied more cheaply and allow for greater control of distributional effects.

While offering clear and stringent analyses of EPI choice, these neoclassical economics and public choice theories have also been criticised for considering only a limited set of instruments, assuming unrealistic utility maximising behaviour, and being overly deductive (see e.g. Howlett and Ramesh 1993, pp. 6-7). However, these theories and their critique will not be considered further in the empirical study of this thesis given the delimitation described above.

3.8.1 Actor resources and interests

As opposed to assuming that economic and political incentives cause certain EPI choice outcomes, some authors have focused on the power resources available to actors involved and the balance of power among them when explaining instrument choice. The process of EPI choice is then seen as a process of political bargaining, in which some interests exert more influence than others[29].

Macdonald (2001) analyses the relative *coerciveness* of instruments used by the Canadian government to address three successive pollution threats over the course of the past century; sewage, industrial toxic waste and greenhouse-gas emissions. He seeks to understand whether the general level of coerciveness has increased or declined over time, and what may have caused this. It is found that relatively coercive instruments were used to address sewage problems from the start, while policy instruments for toxic waste were gradually made more coercive from the 1970s to the 1980s. Climate policy, finally, is the policy area with the least coercive instruments. Thus, it is concluded that "chronologically, we see an overall trend towards declining coerciveness of instruments used but with an instance of increased coerciveness after the instrument was first used" (p. 179). To explain this pattern, Macdonald argues that the degree of environmental risk or the self-interest of bureaucrats to subsidise rather than coerce are not the only determinants of coerciveness.

Instead, Macdonald concludes that "the answer lies in governments' varying ability to do so, which in turn depends on the distribution of power in the policy network" (p. 185). The policy network is here limited to including only the regulator and the regulatee (polluting industry). '*Distribution of power*' is not further conceptualised, but the following factors are referred to: a vital role of the industry in the economy, powerful government friends in the industrial development departments, reliance of the regulator on the expertise and cooperation of firms for data on pollution control, and the degree of organisation among polluters. In the case of sewage, the power of polluters was rather low due to their low level of interest and lack of organisation. For toxic waste, the power of regulators increased in the 1980s as the problem benefited from the 'second wave' of popular support for environmental protection, which led to increased budgets for environment departments. Climate policy, finally, was introduced with less coercive instruments due to the effective organisation of major polluters and decreased budgets of environment departments.

[29] For a study of study of political bargaining at a supranational level (as opposed to the national level which is the focus of this thesis), and more specifically the negotiations on an EU CO2/energy tax, see Klok (2002).

With a similar approach, Daugbjerg has in a series of publications analysed the adoption of EPIs, in particular taxes on nitrate pollution (Daugbjerg 1998b; Daugbjerg 1998a; Daugbjerg 1999) and CO2 emissions (Daugbjerg 2001b; Daugbjerg and Pedersen 2004) in the Nordic countries. Instead of focusing on the degree of coerciveness of EPIs like Macdonald, Daugbjerg argues that the *perceived balance of costs and benefits* is the key feature of EPIs. He claims that the order of preference for instrument types by polluters is based on perceived costs and benefits; negative economic instruments are least preferential, followed by regulatory, voluntary and informative, and positive economic instruments.

Based on a comparative analysis of Swedish and Danish agri-environmental policy instruments aimed at reducing nitrate runoffs, Daugbjerg (1999) finds that Danish policy instruments were more acceptable to farmers than the instruments adopted in Sweden, i.e. the benefit/cost ratio was perceived as more favourable. He argues that technical considerations, the level of knowledge about nitrate pollution, and the nature of nitrate pollution problems cannot explain this outcome. Instead, it is concluded that "this reflects the fact that Danish farmers have more political power than their Swedish counterparts" (p. 163). Interest associations are seen as key actors in his analysis, and they have strong incentives to ensure that members feel that costs are fairly distributed. In comparison to Macdonald, Daugbjerg is more specific in his interpretation of 'political power'; it refers to "whether agriculture [producer interests] has been able to keep the consumers, i.e. its opponents on the market, away from a central position in agricultural policy negotiations" (p. 172). It is demonstrated that the Danish agricultural policy network more successfully excluded consumer interests in the choice of instruments. Accordingly, instruments perceived as more costly to farmers were adopted in Sweden, in particular a universal, negative economic instrument – the fertilizer tax introduced in 1984.

In another article, Daugbjerg (1998a) state that the nature of the policy network is the key independent variable. It is stated that when "a tight and closed policy community, consisting of producers and their traditional negotiating partners/allies within the state, exists with the sector…policy makers tend to introduce low cost environmental policy" (p. 292), i.e. informative instruments and positive economic instruments. Conversely, a loose and open issue network, where conflicting interests are represented, will tend to choose high cost policies. However, the (changing) balance of power in the network is still the decisive factor rather than more stable network characteristics such as cohesion and interconnectedness (see section 3.7.3); "the more structural power the networks provide to farmers, the more likely policy makers are to accommodate their interests in green tax schemes" (Daugbjerg 2001b, p. 105) (cf. Bressers and O'Toole 1998). Daugbjerg's later empirical analysis of the CO2 tax schemes in Sweden, Norway and Denmark also confirmed this proposition (Daugbjerg 2001b).

Both Macdonald and Daugbjerg provide coherent accounts of EPI choice, and, in comparison with some other studies reviewed in this chapter, offer both theoretical and empirical insights. This suggests that examining actor interests and resources at a micro-level is important for understanding EPI choice. However, the power variable seems rather tricky to measure. Macdonald avoids to define it precisely and refers to a wide range of factors when discussing the distribution of power between regulator and regulatee. Daugbjerg, on the other hand, measures it in terms of access to and representation in policy networks.

3.8.2 Individual values and perceptions

A proposal for a micro-level approach to EPI choice not centred upon the economic or political interests of the actors involved, but on individual policy-makers' values and perceptions, has been made by Linder and Peters 1989). They state that "[c]haracteristics of the political system, such as national policy style, the organisational setting of the decision-maker, and the problem situation are all likely to have some influence over the choice of instruments. The relative impact of these variables, however, is likely to be mediated by subjective factors linked to cognition" (p. 35). Hence, they suggest that 'individual perceptions' and 'subjective values' intervene between contextual factors and the choice, and need to be studied further. According to Linder and Peters, "understanding the subjective side of .. instruments should also clarify the various meanings assigned to them, their status, and the logic (or illogic) behind their selection" (p. 55).

This argument is developed in Linder and Peters (1998) where they outline a 'constitutivist' approach to instrument choice. It is argued that within such an approach there is no neutral meaning of instruments, but only subjective meaning; "[it] refers not only to non-instrumental aspects, such as symbolic and ethical import, but also to the instrument features whose interpretation and significance are mediated by values and perceptions" (pp. 40-41). The subjective meaning is shaped by "social or professional interaction as much as by contemplation" (p. 41). However, no concrete suggestions are made as to what these values and perceptions may be, other than that "[w]e expect the conservative policy analyst with extensive training in neoclassical economics to construct different 'matches' [between problem and instrument] than would the reformist political scientist with roots in transformative politics" (p. 42). While this is an innovative approach to the study of instrument choice, it is yet undeveloped theoretically and untested empirically.

3.9 Conclusion

The purpose of the literature review presented in this chapter was to provide a background for addressing the second set of research questions in this thesis; *what determines the instrument choice process? In particular, to what extent does it approximate the procedural rationality ideal?* It started out by demonstrating that the notion of rationality within instrument choice processes can be divided into substantive rationality and procedural rationality. While the former is associated with specific decision criteria and leads to the identification of one particular instrument as the most rational solution, the latter only specifies certain decision-making steps to be followed. It is the latter that is of interest in this thesis given the commitments by policy-makers to strive for this ideal and its potential role in depoliticising EPI choice, as described in chapter 1.

The feasibility of procedural rationality in the EPI choice process has been severely questioned. Much of the critique is aimed at the inherent constraints on rationality in decision-making in general, due to cognitive limitations but also due to the political context. For example, it has been argued that the logical sequence of steps is a misrepresentation and that the process is more iterative. Furthermore, policy objectives are not always clearly and unambiguously defined, leaving the choice of instruments to be the stage where conflicts are played out. A more fundamental critique, though, is that procedural rationality could at best occur in the form of *post-hoc* rationalisation.

Despite these critiques, it was argued that it was still worthwhile to study the extent to which the procedural rationality ideal is achieved, for two reasons. First, the critiques are unnecessarily sweeping and do not recognise differences in the *degree* to which procedural rationality characterises a process. Second, with the recent resurgence of new procedural rationality measures in practice, there is a need to investigate how successful they are. Therefore, an analytical framework breaking down procedural rationality into operational elements will be developed in the next chapter.

However, we also need to understand the political context of procedural rationality. How does it function in a world of power, interests and ideas? For this purpose, a wide range of instrument choice theories were reviewed that highlight the role of political and institutional factors. Some of these examine the adoption of particular instrument types and do not conceive the choice as one between several possible options, like this thesis aims to do. At the *macro-level*, the legal framework constrains the choice (in particular with regards to economic instruments), while the effect of policy style and political culture is less clear in the existing literature. Party ideology and politics, however, have been established as important explanatory factors, in particular with reference to the adoption of environmental taxes. At the *meso-level*, institutional arrangements and organisational culture create path dependence and resistance towards new EPIs. Lately, more attention has been focused on the role of ideas (as such or within international policy transfer) in driving the choice of EPIs, although there is no broad agreement in the literature as to their practical significance. A complex set of instrument choice propositions based on the interconnectedness and cohesion of policy networks have also been put forward, but with limited empirical testing. Finally, at the *micro-level*, it was seen that actor interests and power resources play important roles in the choice of EPIs and the determination of their coerciveness and net costs to target actors. Initial theoretical work has also been presented on a constitutivist approach to instrument choice emphasising the role of subjective values among individual policy-makers.

Overall, there is no one clear message in the literature as to which factors are most important for understanding EPI choice. Indeed, most studies of instrument choice have emphasised the complementarity of factors at the macro-, meso- and micro-level when understanding EPI choice. As described by Linder and Peters (1989), there is a complex ecology of contexts to consider. Therefore, a nested framework in which the key propositions from the macro-, meso- and micro-level literature above are drawn out will be constructed in the next chapter, and subsequently applied to the case studies (chapters 7 and 8). The next chapter will also address the key question of how procedural rationality measures interact with the political and institutional factors identified in the previous sections. How do these factors set the conditions for the procedural rationality ideal? How do these factors shape the use of such measures, for example which decision criteria are set formally and informally? How does procedural rationality function in real world politics?

Chapter 4

Analytical Framework and Methodology

4.1 Introduction

Chapters 2 and 3 discussed the theoretical background of the two research questions investigated in this thesis, namely i) if the instrument mix has become more diverse and ii) what determines the choice of environmental policy instrument. The purpose of this chapter is to develop the analytical frameworks for examining these research questions and to describe the methodology by which empirical data has been collected.

This chapter has been divided into two parts, corresponding to the two research questions. First, the analytical framework and methodology for studying the EPI mix in Sweden and England is described. Second, the framework for studying the EPI choice process is described, followed by a description of case selection and data collection methods. The chapter is then concluded with a critical reflection upon the methods used in this study and suggestions for further methodological development.

4.2 Studying the EPI mix

4.2.1 Analytical framework for the mix of EPIs

To address the first question, there is a need to measure the *diversity* in some way. The simplest way to operationalise diversity is by nominal measurement and categorisation of EPIs in a given mix, to understand the *variety of EPIs,* both within and across categories. The first step is to create an instrument inventory.

For the purpose of identifying individual instruments (and determining what does not count as an instrument), the definition formulated in chapter 2 (section 2.2) is used: *the set of techniques by which governmental authorities wield their power in attempting to ensure support and achieve public policy objectives*. Using this definition means that policy objectives and targets are seen as distinct from instruments and internal organisational arrangements are excluded. Note also that only positive choices of new instruments are considered in this study, i.e. not the removal of an old instrument or the consideration of an instrument that was not adopted in the end. This means that any systematic biases in terms of adopting certain EPIs, while only considering others, are not explored within this study. Furthermore, identifying instruments may involve problems of aggregation, in that it may be unclear what constitutes a single instrument and what measures are sub-components of an instrument. Consistency in the identification exercise is the only way to overcome this problem.

To then categorise instruments into EPI types, the categorisation scheme of the OECD (2001c) is used in this thesis:

- *Command-and-control instruments* – e.g. licenses/permits, ambient quality standards, emission standards, process standards, prohibition bans
- *Economic instruments* – e.g. charges, taxes, tradeable permit systems, subsidies, deposit-refund systems
- *Liability, damage compensation* – e.g. strict liability rules, compensation funds, extended producer responsibility
- *Education and information* – e.g. public campaigns, technology diffusion, eco-labelling, publicity of sanctions for non-compliance
- *Voluntary approaches* – e.g. unilateral commitments, public voluntary schemes, negotiated agreements
- *Management and planning* – e.g. environmental management systems, zoning, land use planning

By plotting instruments into these categories over the time period (1995-2005) in a diagram, the evolving pattern of diversity will emerge. As stated in chapter 1, increasing diversity would be expected, as a consequence of policy-makers' commitments to broaden the national repertoires of EPIs. The results from this exercise are analysed in chapter 6.

Besides from the use of various EPI types, it was found in chapter 2 that the degree of *coerciveness* of an instrument – which can vary both between and within instrument categories – is another relevant aspect to consider when characterising a policy instrument mix. Two different hypotheses regarding the pattern of coerciveness over time were discussed. Doern and Phidd (1983) argued that a pattern of initially minimal coerciveness that gradually increases only as necessary to solve problems could be expected, as a consequence of the liberal democratic ideal. Van der Doelen (1998), on the other hand, argued that one would expect a ‘give-and-take’ pattern of more and less coercive instruments, as a consequence of governments needing both to effectuate policy and legitimate it.

To analyse the pattern of coerciveness of instruments in the mixes, the respective typologies of Doern and Phidd (1983) and van der Doelen (1998) are used and compared in this study. The EPIs identified in the inventory exercise will be slotted into these types and the actual pattern will be compared with the predicted patterns. The original typology of Doern and Phidd presented in chapter 2 is displayed below in **Table 11**. Note that some adaptation is needed to better fit the context of this study. First, the category of ‘private behaviour’ is not relevant here since it falls outside the instrument definition employed here. Second, the instrument examples listed under ‘exhortation’ are activities rather than objects (see de Bruijn and Hufen 1998, pp. 13-14), and thus not considered here. Instead, information campaigns and ecolabelling schemes are considered as instruments based on exhortation. Third, under ‘regulation’ the use of fines and imprisonment as sanctions are consequences of the (non-)compliance with instruments rather than instruments themselves. Fourth, ‘public ownership’ is a question of organisation rather than a policy instrument to achieve a specific objective (see Vedung 1998, p. 38), and thus not considered here. Therefore, these types of instruments are not considered here. Further comments regarding the application of this typology are made in chapter 6 (section 6.3).

Table 11. Doern and Phidd's typology of policy instruments

Private behaviour	Exhortation	Expenditure	Regulation	Public ownership
Self-regulation	Speeches Conferences Advisory investigations	Grants Subsidies Transfers	Taxes Tariffs Fines Imprisonment	Crown-corporations Mixed-corporations
Minimum------------------------------Degree of legitimate coercion------------------------------Maximum				

Source: Adapted from Doern and Phidd (1983, p. 111, figure 5.1).

The original instrument typology of van der Doelen 1998) is displayed in **Table 12**. The only adaptation here is that the ‘repressive’ form of communicative control, i.e. the use of propaganda as a policy instrument, will not be investigated. To make such a judgment of an information instrument would be qualitatively difficult, and require much more investigation than possible within the scope of this study.

Table 12. Van der Doelen's stimulative/repressive typology

	Stimulative	Repressive
Communicative control model	Information	Propaganda
Economic control model	Subsidy	Levy
Judicial control model	Contract/covenant	Order/prohibition

Source: van der Doelen (1998, p. 133, figure 5.1).

These two typologies, the policy instrument definition and EPI categorisation scheme described above thus constitute the analytical framework for the 'mix' part of the study. Below, the research design and data collection methods used to apply this analytical framework are described.

4.2.2 Research design

Overall, a *qualitative, comparative case study design* was chosen for the investigation of the two research questions of this thesis. The *qualitative* approach – in this case consisting of collection and coding of text and interviews – follows naturally from the fact that the wide range of EPIs considered here are unsuitable to advanced measurement and quantitative analysis (see Strauss and Corbin 1998 p. 11; Flick 1998 p. 272). First, as a population of individual observations (i.e. policy instruments), an EPI mix is not sufficiently homogeneous to allow for meaningful statistical analysis. If the research had been limited to a particular and more internally comparable EPI (such as carbon taxes) global statistical analysis of various instrument properties would have been more appropriate (see e.g. Tews, Busch et al. 2003). Still, the simplest form of quantitative analysis, i.e. the measurement of frequencies of instrument types, is employed in this study. Second, research on the policy process of choosing EPIs is most meaningfully undertaken in a qualitative way, by identifying key decision points through document analysis and eliciting stakeholder views and opinions through interviews.

The reason why a *comparative* approach was selected was that "it forces greater specificity on the researcher" (Peters 1998, p. 4), in terms of addressing the root causes of the performance of the political system studied. In this case, this specificity refers to the reasons why procedural rationality was achieved or not, whether it significantly shaped the final instrument choice, and the success in diversifying the resulting EPI mix. According to Peters (1998, p. 10), there are five types of comparatives studies: single country descriptions of politics; analyses of similar processes and institutions in a limited number of countries selected for analytic reasons; studies developing typologies or other forms of classification schemes for countries or subnational units; and statistical or descriptive analyses of data from a subset of the world's countries; statistical analyses of all countries of the world. This study belongs to the second type, focusing mainly on a similar process – the EPI choice process. It can also be described as a 'configurative approach', which means that "the primary purpose is the thorough description of a case or cases, so that the consumer of the research will be capable of comprehending the logic of political life in that limited number of settings" (*ibid*., p. 6). To allow for such thorough description, it was decided that only two countries would be included in the comparison. It was deemed to be important in this study to understand the social, cultural and economic context of EPI choice, i.e. provide a 'thick description' (Geertz 1973, in Peters 1998 p. 6), since EPI choice is not a completely routinised task and can be an arena where many preferences, institutional biases and political cultural factors are played out.

The reason why *Sweden* and the *UK (England)*[30] were selected as countries for comparison were stated in chapter 1 (section 1.5); they have made similar commitments to employ diverse EPI mixes and ensure procedural rationality in the policy process. Importantly, this choice of countries was also made for practical reasons, namely the absence of language barriers, access to data, and previous knowledge of environmental policy in these two countries (see Nilsson and Persson 2003; Persson 2003). The comparative politics literature distinguishes between different logics when selecting cases for a two-country 'causal-analytic' study. The classical account by John Stuart Mill in 1846 identified three conditions under which causation between an independent and a dependent variable could be proved; the method of agreement, the method of difference, and the method of concomitant variation (Ragin 1987, pp. 36-42). Essentially, these methods would allow systematic elimination of possible causal

[30] Note that the devolved administrations in the UK control their own municipal waste policy to a significant extent. Therefore, a choice was made to focus on England, where there were sub-national differences in the UK.

variables. In the contemporary social sciences, however, a more pragmatic approach is proposed due to skepticism of the feasibility and appropriateness of proving causality. Peters (1998, p. 37) states that "[c]omparative politics involves the development of theories explaining behaviour within groups of countries that are essentially similar […] It is also about contrasting cases that are different in any number of ways. Either focus of comparison – explaining similarities or differences – can tell the researcher a great deal about the way in which governments function."

In the context of this study, the similarities between the two country cases thus relate both to the commitment to a diverse EPI mix – which will be examined in the EPI mix study presented in chapter 6 – and to the commitment to increasing the procedural rationality of the environmental policy process – which will be examined in the two instrument case studies in chapters 7 and 8. However, the empirical material could also be viewed from a perspective of difference, in terms of the leader and laggard situation with regards to the successes in tackling the municipal waste problem.

Besides the focus on Sweden and the UK (England), the decision was made to study EPI choice within the policy field of *municipal waste management*. The primary reason was that municipal waste is an interesting and relevant policy field *per se*, as explained in chapter 1 (section 1.5). The waste problem is deteriorating in many ways and has not yet been adequately tackled by existing policies. Policy-makers have emphasised the need for new forms of EPIs in this field. Some delimitations of scope were needed to make the study of waste policy practically feasible, though. First, the empirical analysis in this thesis is delimited to the *municipal waste stream* and does not include hazardous, industrial, agricultural, mining, construction or other key waste streams. Second, the focus is on instruments for preventing or steering municipal waste towards different *waste management options* (prevention, reuse, recycling, incineration and landfill), rather than collection and transport of waste. This means instruments that affect waste management options and their physical availability, legality of use, technical capacity and performance, relative prices and competitiveness, as well as the knowledge about and attitudes towards them.

Another important delimitation refers to the level of analysis. In this study, the focus is on the *national level* of waste policy-making in an EU context. It should be noted that the local level is also significant in waste policy from an instrument perspective. In particular, waste collection charges, recycling schemes and consumer information are important EPIs that are often developed locally. However, the national level was selected here because it is a more strategic level, there is a larger legal mandate (e.g. to set fiscal instruments), and local government is often a key target actor itself for EPIs developed by central government. At the same time, the national level is itself subject to internationally adopted policy, in particular EU legislation and strategies. EU influences on national choices of EPIs will be considered here, but there will not be an examination of the EU institutions as developers of EPIs. Despite the EU legislative activity in the waste field recently, the degree of freedom at the national level is still sufficiently large to cause variation in the instrument mix. To an extent, this inquiry is thus a comparative study of implementation of EU legislation, but the scope will include domestic waste policy instrument initiatives also.

A final delimitation refers to the *temporal scope* of this study, which stretches from 1995 to 2005. A decade provides sufficient time for new policy decisions to be prepared and processed, new policy-making routines to take effect, and changes in the magnitude of the waste problem to be visible. At the same time, it is not excessively long requiring extensive retrieval of historical data.

This study of municipal waste policy could also be used as a basis for generalising findings about EPI choice to other environmental policy areas. To the extent that one wishes to generalise the results, however, it is important to be aware of what the municipal waste policy as a case represents (Flick 1998; Peters 1998). One of the key *problem characteristics* that have been found to influence EPI choice is the *physical and geographical features* of environmental problems, which may limit the range of potential instruments (Weale, Pridham et al. 2000, pp. 139-141). For example, whether an emission problem emanates from a point or non-point source, as well as its measurability and the possibility to attribute it to certain activities or actors, may significantly influence how precise and targeted the EPI can be. Arguably, municipal waste is a controlled and measurable problem compared to environmental problem such as eutrophication and CO2 emissions, implying that rather precise instruments are possible. Secondly, the *availability of technical solutions* is commonly seen as an important factor (Arentsen 1998; Tews, Busch et al. 2003). Municipal waste represents a mature policy field with a range of waste management technologies available, so this should not be a limiting factor upon EPI choice in this policy field. However, there has often been disagreement on the ranking of management options for certain waste materials as well as whether to aim high or low in the hierarchy and how to avoid technological lock-in. Finally, another key problem characteristic is the *visibility and political salience* of the problem. The more risky a problem is perceived to be, the higher need for certainty, precision and effectiveness of instruments (Rittberger and Richardson 2003; Tews, Busch et al. 2003). Although municipal waste is a visible problem in society, it is not generally perceived as risky compared with for example chemicals management and nuclear power (with the exception of waste inicineration which often has higher political salience). Therefore, EPIs giving rise to more imprecise and uncertain outcomes (such as a landfill tax) should generally be more acceptable. Overall, considering these problem characteristics of municipal waste, it appears that EPI choice should not be as limited as it could potentially be for other environmental problems.

Based on this overall research design, it was decided that the study of the policy instrument mix for municipal waste in each country should be complemented by a study of the wider instrument mix used for environmental policy within the two countries. This will provide a context for the understanding of trends within the waste policy field. The delimitations of this overall EPI mix study were set as a consequence of data availability (see below).

4.2.3 Data collection

With the analytical framework and research design in place, the first step was to do an overview of the waste policy framework in each country over the time period studied, including the key features of the waste problem, historical policy approaches, the growing influence of EU waste legislation, and the main policy strategies and targets introduced during the study period. This overview is presented in chapter 5. With this backdrop, sources of data on policy instruments adopted in the municipal waste field in the period 1995-2005 were identified, as well as sources of data on EPIs more broadly.

For the compilation of *waste policy instruments*, there was no existing source that was sufficiently complete for either Sweden or England. A collection of different documents and websites was therefore reviewed; policy bills and strategies, websites of relevant government departments and authorities, and reports by the EEA and OECD. The key sources are listed in **Table 13**. In addition to these primary data sources, some secondary sources were used in the case of the UK (e.g. Davoudi 2001; Bulkeley, Watson et al. 2005; Davoudi and Evans 2005). A general problem was that there was much fewer academic studies of Swedish waste policy from a political or economic point of view to draw on.

Table 13. Sources of information on waste policy instruments

Sweden	England
Waste policy bills and strategies • 1993 Waste Bill (Regeringens proposition 1993) • 1997 Waste Bill (Regeringens proposition 1997) • 2003 Waste Bill (Regeringens proposition 2003) • 2005 Environmental Objectives Bill (Regeringens proposition 2005a) • 2005 Waste Strategy (Naturvårdsverket 2005) *Waste policy reports* • Inquiry review of Swedish waste policy (Statens Offentliga Utredningar 2005a) *Government websites* • Ministry of Sustainable Development (http://www.regeringen.se) • Swedish EPA (http://www.naturvardsverket.se) *Other* • OECD Environmental Performance Review of Sweden (OECD 2004b) • EEA report on implementation of Landfill Directive(EEA 2002) • Annual reports of the Swedish Association of Waste Management (RVF) (RVF 2003b; RVF 2004; RVF 2005b; RVF 2006c)	*Waste policy bills and strategies* • 1995 Waste Strategy (DoE 1995) • 1999 Draft Waste Strategy (DETR 1999b) • 2000 Waste Strategy (DETR 2000a) • 2002 Waste Strategy Review (Strategy Unit 2002) *Waste policy reports* • Follow-up on the Waste Implementation Programme (DEFRA 2005g) *Government websites* • Department for Environment, Food, Rural Affairs (DEFRA) (http://www.defra.gov.uk/environment/waste/index.htm) • Department of Trade and Industry (DTI) (http://www.dti.gov.uk) • Environment Agency for England and Wales (EA) (http://www.environment-agency.gov.uk/) • Waste and Resources Action Programme (WRAP) (http://www.wrap.org.uk/) • The Landfill Tax Credit Scheme (LTCS) (http://www.ltcs.org.uk/) *Other* • OECD Environmental Performance Review of the UK (OECD 2002) • EEA report on implementation of Landfill Directive (EEA 2002) • Green Alliance waste policy review (Green Alliance 2002)

The first step was to review the latest major waste policy bills and strategies in each country and record the instruments referred to, based on the EPI definition and categorisation scheme described above. Then, the various other sources in **Table 13** were used to double-check that all instruments were included and to avoid double-counting (note that some instruments were referred to by different names in different sources). In several cases, more specific sources of information on individual instruments were used, and these are referred to in the analysis of the results in chapter 6.

Also for the overview of *EPIs used within environmental policy generally,* multiple sources had to be used. Ideally, there would be internationally comparable databases of the EPIs used in each country, including all the six EPI categories. This would enable a meaningful assessment of which EPI types are generally preferred. However, such databases do not exist, and alternative sources had to be explored. First, the possibility of reviewing major environmental policy bills was looked into, but the significant differences between the content, purpose and frequency of bills between Sweden and the UK prevented such an approach. Scoping was also made of national reports submitted for Agenda 21 follow-up and to the UN Framework Convention on Climate Change as potential data sources. However, they were not found to be sufficiently complete to merit a more detailed analysis.

Instead, it was decided that the broader EPI overview should be based on a collection of international and national sources, that focused on particular EPI categories or types of instruments. These sources are listed in **Table 14**. Note that for this overview, UK-wide EPIs were considered. Most data is available on the use of economic instruments (in particular taxes and charges) and voluntary approaches, possibly because these are perceived to be more discrete and separate instruments than for example command-and-control regulations and management and planning instruments. There has also been a strong interest in recording the existence and experience of using such novel EPIs from the policy-maker community.

Table 14. Sources on information on EPIs in Sweden and the UK

International EPI overviews

- The OECD on-line access database on *Environmentally Related Taxes* (http://www.oecd.org/document/29/0,2340,en_2649_37465_1894685_1_1_1_37465,00.html) (OECD 2003b), which has since been subsumed into the OECD/EEA on-line database on *Economic instruments and voluntary approaches used in environmental policy and natural resources management* (http://www2.oecd.org/ecoinst/queries/index.htm) (OECD 2003c)
- OECD Environmental Performance Reviews of Sweden (OECD 2004b) and the UK (OECD 2002)
- The EEA review of environmental taxes in EU member states conducted in 2001 (EEA 2001) and in 2005 (EEA 2005), and the review of market-based instruments in 2006 (EEA 2006)

National EPI overviews

Sources on Sweden:

- OECD Environmental Performance Review of Sweden (OECD 2004b)
- National environmental accounts, including lists of environmental taxes (SCB 2005) and subsidies (SCB 2006)
- A survey of VAs used in Sweden conducted by the Swedish EPA in 2000 (Naturvårdsverket 2000b)
- A survey of economic instruments conducted by the Swedish EPA in 2003 (Naturvårdsverket 2003)
- A survey of economic instruments conducted by the Swedish EPA and Swedish Energy Authority in 2006 (Naturvårdsverket & Energimyndigheten 2006)

Sources on the UK:

- OECD Environmental Performance Review of the UK (OECD 2002)
- National environmental accounts, including list of taxes and charges (National Statistics 2006; Gazley 2006)
- Jordan, Wurzel et al. (2003e) overview of NEPIs used in the UK
- Work on a database of DEFRA regulations was initiated for the 2004 Regulation Taskforce Report (DEFRA Regulation Taskforce 2004). When contacted, staff at DEFRA explained that it was not complete and would not be publicly accessible in the near future.

The point of departure for recording the use of *economic instruments* and *voluntary agreements* (*VAs)* in this study were two international databases created and maintained by the OECD and EEA, which have since the initial query been merged into one: *Economic instruments and voluntary approaches used in environmental policy and natural resources management* [31]. The database was initially based on a 1998 survey and the national contact persons have since been invited to update the information. The benefit of this database is that instruments are entered in a consistent way by the national contact persons, but it is clearly stated that it is not complete for all countries

[31] The initial query of the database for this study was performed on 1 July 2004. By then the Swedish data had not been updated since 1998, while the UK data were updated in the winter of 2003. A second query was performed on 6 January 2007 to verify the information. By then the Swedish data on taxes and subsidies had been updated, to include rules applying on 1 January 2006. For the UK, data on taxes had also been updated to describe rules applying on 1 January 2006. Other instrument categories were still incomplete. The verification exercise of the initial database results did not result in any major changes of the list presented in Appendix IV.

and policy areas, and that much of the data is obsolete. These weaknesses were confirmed as the long list of instruments generated from this database was double-checked with national data sources available in Sweden and the UK. These national sources included national environmental accounts, OECD Environmental Performance Reviews, and reports by the Swedish EPA (see **Table 14**). This double-checking exercise led to the addition of some instruments, removal of some out-of-date instruments, and re-categorisation of some instruments according to this study's scheme. Some local-level instruments reported in the sources were excluded, such as household water charges and waste collection charges, in line with the national-level policy focus of this thesis. The final inventory of economic instruments (including the sub-categories of taxes and charges, tradeable permit systems, deposit-refund systems, and subsidies and grants) and VAs is presented and discussed in chapter 6.

Regarding *environmental taxes*, the official national environmental accounts were useful for verifying the information from the OECD/EEA database. However, these are still partially inconsistent in the definition of what counts as an environmental tax (see Gazley 2006; OECD 2006 p. 26). These problems will be addressed in chapter 6. The use of *environmental charges* is more difficult to examine, since such information is not routinely collected through a standardised methodology. Therefore, it was not possible within the scope of this study to verify the information provided by the sources above, through for example contacting authorities in each country responsible for administering some kind of environmental charge.

There was also a lack of up-to-date and complete information on the use of *subsidies and grants*, especially since this EPI category includes diverse instruments such as one-off investment grant programmes, continuous price subsidies, and R&D grants. Therefore, those that are listed in the inventory should be seen as indicative only for the kinds of subsidies adopted. For Sweden, the most up-to-date list was the one compiled by the Swedish EPA in 2003. This list was double-checked with the OECD database and Statistics Sweden's national environmental accounts. For the UK, the OECD/EEA database was used. Based on this list, web-based searches were performed on inidivual schemes to retrieve more information. However, there was a surprising lack of easily accessible data on environmental subsidies in the UK. To mitigate against these data problems, aggregate figures on public expenditure on environmental protection will be reviewed in chapter 6 as a supplementary indicator.

For *voluntary approaches*, the decision was made to only include negotiated agreements, and not unilateral commitments or public voluntary schemes (see OECD 2003e). The OECD/EEA database was used as a point of departure, but its incompleteness led to the use of secondary sources; a Swedish EPA report (Naturvårdsverket 2000b) and Jordan, Wurzel et al.'s (2003e) study of the UK.

As for characterising the use of *command-and-control regulation*, there was a lack of a similar international database. In the UK, the DEFRA Regulation Taskforce had initiated work in 2003 on compiling a database of regulations within its areas of responsibility for the regulation review published in 2004 (DEFRA Regulation Taskforce 2004). However, upon request DEFRA communicated that the database had not been completed and it was not publicly accessible[32]. Still, the existing information published in the report could be used for some simple quantitative measures. As for Sweden, the Swedish EPA maintains a list of all its regulations in the environmental field according to the year they were issued, which could allow some analysis of changes over time (Naturvårdsverket 2006a). However, this list excludes older regulations that have ceased to apply or been replaced, so trends over time cannot be identified with certainty. Furthermore, these regulations represent only one form of secondary regulation and not the full body of environmental regulation[33].

The other three EPI categories examined in this study were more challenging in terms of data availability. As for *education and information* instruments, some secondary sources were used for a brief review although no nominal measurement could be performed. An analysis of *liability* instruments, on the other hand, was not included due to the difficulties involved in measuring such instruments in a meaningful way. Furthermore, an overview of liability regimes in the EU suggested that Sweden and the UK are not using them as EPIs to any great extent compared with other European countries, such as Denmark (see EEA 2006). Neither was an inventory of *management*

[32] Email communication with Ms. J Summers, 28 April 2004.
[33] In Sweden, national laws and regulations are issued at three different political-administrative levels. The parliament issues *laws*. The government issues *ordinances*. The central government authorities, like the Swedish EPA, issues *regulations*.

and planning instruments made. The rationale was that it is too difficult to obtain meaningful nominal measures of these instruments, that their use has not changed as much over time as some other EPIs, and that they are often a local matter rather than part of national environmental policy.

To sum up, a number of methodological and data difficulties in compiling an inventory of an EPI mix for a certain policy field mean that the reliability of the results is limited in some ways. Most importantly, there is a problem of incomplete and obsolete data on EPIs, especially data that include and compare different EPI types. As described above, this problem was tackled by identifying and drawing on a variety of complementary international and national data sources. More difficult to overcome, however, is the risk of overestimating the role of economic instruments and voluntary approaches in the EPI mixes due to the bias of attention to these EPI types. This attention bias could be overcome through more primary data collection on EPI types, such as surveys addressed to relevant government departments and authorities, but this was not possible within the scope of the study.

Secondly, the data sources that do exist often use inconsistent definitions and categories for instruments. This problem was addressed through using the definition and categorisation scheme developed in this study (see section 4.3.1), double-checking sources against each other, and thus shortening the initial ‘long-lists’ of EPIs. A related issue is that instruments are often interlinked. For example command-and-control regulations are the basis of many economic instruments, a grant scheme may be related to the uptake of a VA, and payments made within a tradeable permits scheme may be seen as taxes. However, a detailed analysis of the interlinkages between instruments, including synergies and counterproductive effects, was not the aim of this study, which focuses on nominal measurement of EPI categories.

Finally, it should be emphasised that the identification of EPIs depends on the delimitation of the policy area. Many policy instruments – purposely or as a side effect – cut across policy areas. For example, the use of waste incineration is strongly affected by the energy tax system. However, delimitations were needed to make the study practically feasible, even though they may not reflect important policy drivers upon waste management in other sector.

Despite these methodological challenges, studies of the complete EPI mixes are needed to complement more focused studies of the use of a particular EPI type. The aim of this study is to take an initial step towards a methodology for measuring and characterising the full EPI mix affecting a given policy field. Based on the findings of this study, measures to understand to relative significance of different EPIs (e.g. in terms of environmental effect, cost, perceived importance) and the interlinkages between them could be further developed.

4.3 The process of choosing EPI

4.3.1 Analytical framework for the EPI choice process

Moving on to the second research question on what determines EPI choice, it was argued in chapter 3 that it is relevant to consider the extent to which procedural rationality is achieved in the process. A theoretical case for examining procedural rationality was made, and it was also stated that recent empirical trends suggest that there may be a resurgence of procedural rationality in the environmental policy process.

The strongest and most concrete example of such trends is arguably the increased emphasis in the late 1990s and early 2000s on the need to undertake formal *Regulatory Impact Assessment* (RIA) (see OECD 1997; Persson 2003; Malyshev 2006). RIA has been defined as a method of “i) systematically and consistently examining selected potential impacts arising from government action and ii) communicating the information to decision-makers” (OECD 1997, p. 14). Comparing alternative instrument options (whether they be regulatory, economic or information-based) is also an integral part of RIA. In the UK, RIA has been strongly promoted under the Better Regulation initiative and there are now signs of high compliance in government departments (National Audit Office 2006b; Russel and Jordan 2007). In Sweden, there is also a requirement to undertake RIA at the three levels involved in policy-making; the Committee of Inquiry level (Statsrådsberedningen 2000), the Government Offices level (Regeringens skrivelse 1997), and the central government authority level (Ekonomistyrningsverket 2003). As described in chapter 1, there has also been a new Impact Assessment (IA) procedure initiated in the European Commission.

Whereas RIA is a concrete expression of the procedural rationality ideal, it could also be seen as part of a broader trend of *professionalisation of policy-making*. As mentioned, the UK Cabinet Office has argued that policy needs to be more strategic, outcomes-focused, forward-looking, creative, innovative, robust, and joined-up (Cabinet Office 1999). More strategic policy means that alternative policy instruments need to be considered properly. In comparison, Sweden has been seen as a country with traditionally rationalistic, knowledge-based policy-making, not least manifested through the regular use of commissions of inquiry for investigating new policy issues (Lundqvist 2004).

Another broader trend in which the quest for new EPIs could be seen is the promotion of *'evidence-based policy'* (Solesbury 2001). Learning more from the past through structured evaluations and drawing lessons from academic research has been emphasised strongly in British policy-making the last decade, although more so in the social and health policy fields.

These empirical trends thus emphasise the need to re-examine the degree of procedural rationality in the EPI choice process. In chapter 3, it was explained that there was a need to unpack procedural rationality into more operational terms and construct an analytical framework that can be used in case studies of EPI processes. In **Table 15**, such a framework of key elements of procedural rationality is proposed, based on the theoretical review in chapter 3 and the empirical trends referred to above. Below, the inclusion of each of the elements will be explained and justified.

Table 15. Key elements of procedural rationality

Element of procedural rationality	Description
Linear sequence of decision-making stages	Was the policy process undertaken in the following order with a satisfactory degree of agreement among parties involved at each stage: definition of problem; setting of objectives; specification of choice criteria; identification of possible instruments; impact assessment of identified instruments; decision? Were strategic decisions regarding the direction of policy and type of instrument taken before operational decisions on detailed design issues? Was the process problem-led?
Clear policy objective and choice criteria	Was the objective unambiguous and quantified? Were the choice criteria mutually exclusive and prioritised?
Several alternative instruments identified	Was more than one possible instrument defined? Were removal of instruments considered? Were interactions with existing instruments considered?
Regulatory Impact Assessment	Were the costs and benefits of each alternative instrument described, quantified, and monetised? Was the distribution of costs and benefits described and quantified?
Consultation	Throughout the instrument choice process, were relevant parties consulted to a satisfactory extent?
Use of evidence	Throughout the instrument choice process, were relevant analyses prepared and used to inform the choice?

At the core of procedural rationality is the notion of a *linear and logical sequence of decision-making stages*, namely; definition of problem, setting of objectives, specification of choice criteria, identification of possible instruments, impact assessment of identified instruments, and decision (see e.g. Howlett and Ramesh 1995 p. 140; Rydin 2003, DeLeon 1999). In addition to being derived from theory, this kind of sequence (and variations thereof) has been emphasised in policy guidance (see e.g. Cabinet Office 1999) and is also implied by the RIA procedure. An analysis of actual EPI choice processes should therefore examine whether such a sequence has occurred or not. As explained in section 3.3 there has been substantial critique of whether this kind of sequence is attainable in practice. For example, a choice of instrument may *de facto* be made before the problem is defined and the objective is specified.

A second core element is the *clarity of policy objectives and criteria*. It was seen in the literature review in chapter 3 that objectives and criteria as a basis for identifying possible instruments and assessing their suitability can be – both deliberately and indeliberately – imprecise, ambiguous, conflicting, and not prioritised in relation to each other (see Majone 1989). Therefore, it is relevant to empirically examine whether the clarity, or lack thereof, promoted or hindered the consideration of alternative EPIs, and if the final choice of EPI is consistent with the prioritised objectives and criteria.

Thirdly, the *consideration of more than one alternative EPI* is an integral element of procedural rationality. In practice it has been found that often surprisingly few alternatives are identified in environmental decision-making (National Audit Office 2006b; Persson 2003). Even when a set of alternative EPIs are identified, they may not be adequately elaborated to provide for a thorough *impact assessment* against objectives and criteria, which is a fourth key element of procedural rationality. As explained above, formal and compulsory RIA procedures are now in place in both Sweden and the UK. These stipulate that a comprehensive range of costs and benefits should be measured, if not in monetary terms, then in qualitative terms.

A fifth element has here been defined as the use of timely and comprehensive *consultation* rounds. Formal consultation is a required part of the RIA procedure in the UK and a key feature of the Swedish policy preparation model. Beside from representing good democratic practice, consultation has been recognised as a source of information and a means to prepare and build acceptance among societal actors for future policy instruments. In this capacity, it can enhance the procedural rationality.

Finally, the *use of evidence* is a key element of procedural rationality. It should take place throughout the process; informing a rigorous analysis of the problem to be solved, drawing on lessons learned when considering options, and modelling possible impacts of different instrument options. In practical terms, this may involve commissioning consultant reports, referring to *ex post* evaluations, and/or considering international experiences.

By applying this framework to case studies of EPI choice, the question whether procedural rationality was achieved in the process or not is first addressed. In a comparative setting, the answer can be expressed as a matter of degree. Second, it will also enable a discussion of whether the procedural rationality elements proposed above helped shape the EPI finally chosen. In other words, can the instrument finally chosen be derived from the comprehensive and systematic analysis of alternative instruments as stipulated by the procedural rationality ideal?

As argued in chapter 3, the literature on policy instrument choice has largely rejected the applicability of the procedural rationality ideal to real world policy-making and the notion of a free EPI ‘choice’, and developed a wide range of other theories highlighting various political and institutional factors shaping the process and likely preferences of different actors. While broad-brushed rejection may be misplaced in the light of recent empirical trends, the procedural rationality measures identified above are nevertheless applied in a world of power, interests and ideas. This decision-making ideal must thus co-exist with these less predictable political pressures and driving forces (Brunsson 2006). The key argument in this thesis is that, considering the real world of policy-making, we should not expect rationality *of* the EPI choice process but we could expect a degree of procedural rationality *within* the EPI choice process. Whether procedural rationality then actively informs the EPI choice or simply amounts to *post-hoc* rationalisation is another question, but, as argued in chapter 3, perhaps a less critical one.

The task for the case studies is thus to understand the role of political and institutional factors on EPI choice, and how they interact with efforts to ensure procedural rationality in the process. A comprehensive literature review of such factors influencing EPI choice was presented in the previous chapter. This thesis will use these insights to examine what circumstances and factors may have shaped EPI choice other than the application of (or lack of) procedural rationality elements. Which other factors can explain the final choice of EPI? Furthermore, how can these factors explain if and why procedural rationality was achieved or not? For the purpose of examining these political and institutional factors, the main findings from the literature review in chapter 3 have been summarised in **Table 16**. In this table the nature of the influence of the macro-, meso- and micro-level factors is described (middle column) and the main predictions regarding specific EPIs have been distilled (right column). This table will be used as a basis to discuss the EPI choice case studies in chapters 7 and 8.

Table 16. Macro, meso and micro level factors influencing EPI choice

	Nature of influence on instrument choice	Specific EPIs preferred (type and/or design features)
MACRO-LEVEL FACTORS		
Legal framework	The overarching national and international legal frameworks influence which instruments are seen as more feasible than others.	• EU state support rules makes it difficult for states to adopt **subsidies and taxes** (with exemptions). • International trade rules makes it difficult for states to adopt **certain labelling schemes, taxes, procurement policies**.
Policy style and political culture	Elements of policy style – such as anticipatory/reactive problem solving, consensual/impositional relationships with organised groups, legalistic/managerial public management style – influence the stringency/flexibility of regulation. 'Statist' countries accept more intrusive instruments and centralised government intervention. Corporatist countries are more conducive to ecological modernisation agenda, hence new forms of EPIs.	• A legalistic public management style is conducive to **command-and-control regulation**, while a managerial style is conducive to more **complex** and **discretionary** instruments. • A consultative style is more likely to result in **open framework regulations** rather than stringent command-and-control regulation. • More **intrusive** and **centralised** instruments are accepted in more 'statist' countries. • Higher use of **economic and informative instruments** is likely in corporatist countries, since they are more conducive to the ecological modernisation agenda.
Party politics and ideology	Political ideologies, and the influence they have given the parliamentary composition, determine which EPIs are preferred.	• Neo-liberal ideas about deregulation increase the uptake of '**NEPIs**', in particular 'soft' instruments. • Left-of-centre parties are more likely to support **green taxation** than centre and right-of-centre parties, which prefer softer instruments. • Centre-left governments increase the uptake of **economic instruments**, due to their support of 'big government'. • Green party presence helps tip the balance towards '**NEPIs**'.
MESO-LEVEL FACTORS		
Institutional arrangements and organisational culture	The institutional embeddedness of solutions to problems and the existence of an organisational culture lead to path dependency in the use of instruments. Professional backgrounds of policy makers in a policy domain matters.	• Because institutions invest time and resources in adapting to a particular EPI, they tend to prefer **not introducing new types**. • Because of the logic of appropriateness, actors prefer **instruments that fit with the existing institutional framework**. • **Instruments used when an organisation was formed** tend to stick. • The professional background matters in that economists tend to prefer **economic instruments**, lawyers **legal instruments**, and engineers **planning instruments**.
Ideas, learning and transfer	General ideas about various instrument types influence the choice in a given situation, and such ideas can result from successful advocacy coalitions, learning processes or international policy transfer.	• During normal periods existing instruments are only **fine-tuned**, while during paradigmatic change **new forms of instruments** are considered. • If instrument ideas figure prominently on the global agenda and diffusion is institutionalised in an international organisation, **policy instrument innovations** are more likely at the national level.

Character of stable policy networks and subsystems	Policy subsystem complexity matters. The network characteristics interconnectedness and cohesion matter for a range of instrument properties.	• With high policy subsystem complexity **market and voluntary instruments** tend to be preferred, while with low policy subsystem complexity **regulatory or mixed instruments** tend to be preferred. • The more cohesive a policy network is, the less **normative appeal** is needed, the larger tendency for **net provision of resources**, and the more **freedom for target groups to opt out**. • The more interconnected a policy network is, the more **bilateral arrangements** are possible.
MICRO-LEVEL FACTORS		
Actor resources and interests	The more powerful the target group, the more favourable benefit/cost ratio for them and the less coercive instruments.	• The more powerful and organised producer group, the **less coercive** instruments are adopted. • Producers prefer (in descending order) **positive economic instruments, voluntary and informative instruments, regulatory instruments, and negative economic instruments,** out of cost-benefit ratio concerns. The more powerful they are, the more successful they will be in achieving their first preferences.

4.3.2 Research design

As described above (section 4.2.2), a qualitative, comparative research design examining municipal waste policy in Sweden and UK (England) was selected for this study. Within these parameters, the question was how to design an empirical study of the *EPI choice process*, addressing the questions and propositions stated in the analytical framework above. A number of alternative designs are possible. For example, interviews or questionnaires could be conducted with key municipal waste policy-makers, with respect to the normal process of choosing EPIs. However, considering that developing new EPIs is not a daily or routinised task, it is likely that there exists no 'normal' process and there is a risk that the results would be of an overly generic character. Another approach could be to take all or a number of the EPIs for waste identified in the national mixes as observations, and conduct 'backwards tracing' of the processes preceding each one. This approach would be time-consuming and could risk trading off too much detail against the large number of observations. An alternative could be to take all RIAs performed in the field of municipal waste as the units of analysis and examine their consideration of alternative EPIs. Such a study of formal documents would risk ignoring the political and institutional context of the decisions. Instead, a *case study approach* was selected. Studying one EPI choice process in each country allows an in-depth understanding of the political and institutional context, and comparative cases provide more robust results than just a single case study.

In the comparative politics literature, case study designs have been found to work well, due to the common situation of 'many variables, small *n*' or the existence of 'causal complexes' (Ragin 1987, p. 49). Particularly useful for policy process studies, Peters (1998, p. 141) argues that "the case-study can look directly at the sequence of events that produced an outcome, rather than just at the outcome". He finds that, despite the importance of process and procedure in the political system, studies of processes are rather rare in political science. Also recognising the condition of many variables to consider, Yin (1994, p. 13) recommends case studies when there are more variables of interest than data points and the examination is of a contemporary phenomenon within a real-life context, where boundaries between phenomenon and context are not clearly evident, thus leaving the researcher with little control.

The case study literature suggests that *prior development of theoretical propositions and questions* is useful for guiding data collection and analysis (see Yin 1994). This enables more focused use of the case study. However, it has also been seen as advantageous that case study research enables a dialogue between ideas and data (Ragin 1987). On the use of theory in case study research, several different purposes have been identified (Eckstein 1975, in Peters 1998 pp. 147-150); purely factual case studies (configurative-idiographic), looking for the typical (heuristic), illustrating a general hypothesis (disciplined-configurative), testing of a proto-theory (plausibility probing), and testing theory (crucial case). Using this terminology, the purpose of using case studies within this thesis is to look for the typical with regards to the extent of procedural rationality in the EPI choice processes, and illustrate the previously generated general hypotheses regarding the instrument choice process which have been summarised in the macro-meso-micro-framework. This means that the purpose here is to interpret reality, rather than to develop rigorous scientific generalisations (see Peters 1998 p. 173).

In all case study research, criteria are needed for the *selection of cases*. These should stem from the theory used, but it can also be difficult to know *a priori* what a case is a case *of* (Peters 1998, p. 145). This means that some flexibility is required both regarding the formulation of theory and the definition of the empirical case. Nevertheless, the main guideline for using two-cases and comparative designs is that each case should be selected so that it either predicts similar results (literal replication) or produces contrasting results but for predictable reasons (theoretical replication) (Yin 1994, p. 46). In this study – and as described above in the justification of the overarching comparative approach to this study (section 4.2.2) – cases of EPI choice processes were selected to predict similar results, namely that they should have been conducted under the general commitment to a diverse EPI mix and procedural rationality made by both Swedish and UK policy-makers. One would thus expect an *active search for new EPIs* and *procedural rationality* in these processes.

In practical terms, this meant that the processes had to be relatively recent or on-going so that renewed emphasis on procedural rationality had been given time to take effect. The need for recently completed or near-finishing processes was also related to data access, i.e. availability of documents and individuals with good recollection of the process. Finally, there was a need to select relatively discrete processes, i.e. where a coherent and delimited series of documents, debates and decisions could be located. Setting boundaries would be more difficult if the EPI output was a package of instruments resulting from a broad and long-standing debate.

Based on these criteria, one instrument choice process in each country was selected after initial scoping in the spring of 2004. In England, the *landfill allowance trading scheme* (LATS) instrument was chosen. It is a tradable permit system (TPS) for the right to send municipal waste to landfill for local authorities. By spring 2004, the Act introducing its legal foundation had been given Royal Assent and the plan was for it to enter into force in the spring 2005, which it eventually did (1 April). It was thus an on-going but nearly finished EPI choice process, with indications of good data access. The process appeared to be discrete, with a series of three consultation papers leading up to the final design and the instrument was a response to a specific problem; the need to reduce the use of landfills according to the EU Landfill Directive.

In Sweden, the *waste incineration tax* was chosen. The purpose of this tax is to stimulate material recycling and biological treatment, and limit the expansion of incineration capacity. An commission of inquiry with the aim to develop a concrete proposal and legislation had started in 2003 and the original plan was to report in June 2004. The outlook was that a bill would follow, given sufficient support for a new tax from the commission of inquiry and consultees. In the end, the inquiry report was delayed until March 2005. A parliamentary bill was issued in March 2006, resulting in the tax entering into force on 1 July 2006. Good data access could be ensured from the start, since access to the commission of inquiry's internal meetings and working documents was provided. This process was also relatively discrete in nature, in that a few inquiries had been made into its possible design and it responded to a limited (although disputed) waste problem.

The case selection thus resulted in two economic instruments, an environmental tax in Sweden and a tradable permit system (TPS) in England. Although the cases were not selected on the basis that they were NEPIs or economic instruments, it turned out to be helpful that they were easily identifiable (compared with for example a more comprehensive command-and-control regulation) and had clear purposes (compared with for example information measures). Both instruments were also perceived to involve significant changes for the waste management sector in each country, and thus represented important additions to the EPI mixes. In a future study, it would be interesting to compare these two instrument choice processes with ones that did not result in economic instruments being proposed, but other categories.

4.3.3 Data collection

According to Yin (1994, p. 80), six sources of evidence are possible in case study research: documentation, archival records, interviews, direct observation, participant observation, and physical artifacts. For this study, documentation and interviews were used, as well as some direct observation in the the Swedish case study. All the evidence was collected and summarised throughout the research process in case study reports. The first step was to conduct an initial document analysis to trace the policy process backwards; i.e. understand the current decision-making stage, identify key historical and future decision points, identify key actors and institutions involved, and understand the technicalities of the waste problem that was addressed. Mapping out the process in this way ensured that the ensuing detailed document analyses and semi-structured interviews could be used to 'fill gaps' and be focused on the most critical or unclear issues. The goal of the data collection was both to produce a factual description of what decisions were made and to capture the variety of views, opinions, arguments and contextual factors that influenced the final EPI choice. As with much qualitative research, the aim was not to measure the frequency of a certain opinion in documents and interviews but to explore the range of opinions (Gaskell 2002, p. 41).

For the analysis of *documents*, a wide range of written material were collected from different sources for each case, see lists of key documents in **Appendix I**. The *bills and regulations* formally proposing, justifying and specifying the final instruments were collected, including relevant parliamentary records. Of key importance for understanding the process leading up to the choice, were the series of *consultation papers and inquiry reports* that proposed instrument types and designs for stakeholders to consider. In the UK, there was a series of three government consultation papers on what was to become the LATS. In Sweden, there were two commissions of inquiry that had considered the issue of an incineration tax. A few technical *consultant reports* had been commissioned in both countries, either before a consultation paper was prepared or within the commission of inquiry's work. Summaries of and individual *consultation responses* were collected. Together with the consultation papers and inquiry reports, these documents were essential for unveiling the main issues of disagreement, what instruments were officially considered and proposed by stakeholders along the way, and what objectives and criteria were specified. In addition, some *ad hoc* position statements and debate articles by stakeholders involved were collected and websites of relevant organisations and political parties were reviewed. Finally, *press releases and news articles* were collected.

Altogether, a rich set of written material was collected for each case study, but there was some asymmetry in data access between the two countries. In the UK, the responses to the first consultation paper in 1999 were not accessible. Upon request, DEFRA officials stated that those documents were no longer stored in the archives[34]. For the latter two consultation papers, summaries of the outcome were available and some invidual responses from key stakeholders were retrieved. The lack of access to the early consultation responses was a problem, though, since that was the stage of strategic decisions on the EPI, as opposed to technical details. Useful substitutes for these responses were eventually found in the written and oral evidence given by the same stakeholders to parliamentary inquiries on landfill and waste policy. This evidence

[34] E-mail communication with Ms. Faith Opio and Mr. Mike Scanlon in October 2005.

communicated views and opinions by the actors, including the government itself, that could not always be found in other statements. Another problem with data access in the UK was encountered when a key consultant report estimating the costs and benefits of the proposed instrument was located but not released[35]. Overall, though, the accumulated body of documents provided ample information on the EPI choice process.

In Sweden, access to formal and informal documentation was not a problem. Instead, access to internal working documents of the 2003 commission of inquiry was provided. These included draft chapters of the final report, consultant reports commissioned for the final report, submissions by individual stakeholders, and email communication between the members of the commission. Normally, such working material is not publicly accessible in Sweden.

For the *interviews*, it was decided that they should be semi-structured in order to collect comparable responses but still allow interviewees to elaborate on various points (see Fielding and Thomas 2001, p. 124). Furthermore, it was clear that they would be a form of élite interviews, since the number of people and organisations involved in national waste policy-making is limited and they are highly specialised experts. There were thus no issues of sampling involved in the selection of interviewees. According to Kvale (1996, p. 101), "interviews with elites… involve problems of access to the interviewees, and generally require that the interviewer has a good grasp of the interview topic in order to entertain an informed conversation".

Regarding the *selection of and access to interviewees*, potential interviewees were first identified in the policy documents and stakeholders' position statements. Upon contact, some of the interviewees also referred to other people involved. In parallel with the 'snowballing' effect, efforts were made to ensure that the key interests would be represented in the interviews – i.e. central government, local government, waste management industry, manufacturing industry, and environmental agencies and organisations – in order to capture the variety of opinions (rather than relative frequency, see also Dexter 2006 p. 19). The list of interviewees for each case study is displayed in **Appendix II**.

[35] Upon repeated requests, Enviros consultants did not release the set of studies modelling the economics of a landfill TPS, despite written consent from DEFRA.

In Sweden, a total of 16 interviews were conducted. The selection of interviewees was facilitated by the fact that most key experts and stakeholder representatives were also members of the 2003 commission of inquiry into an incineration tax. An initial contact with interviewees could thus be made during the first commission meetings attended in the spring 2004. They all subsequently agreed to be interviewed with exception of the chairperson who declined based on time constraints. The interviewees represented the government ministries involved (Finance, Industry, and Environment), government agencies (the Swedish EPA and National Tax Board), the waste management industry (the Swedish Association of Waste Management and the Recyling Industries' Association), the manufacturing industry (the Confederation of Swedish Enterprise), academia, and the commission of inquiry secretariat. One interviewee was later added, namely the desk officer for waste at the Swedish Society for Nature Conservation which is Sweden's largest environmental NGO.

In the UK, a total of 15 interviews were conducted. Initial contact was made with DEFRA and their relevant members of staff were interviewed. Based on their information and the policy documents, names of other relevant organisations and people were compiled. In the end, the interviewees represented the Environment Agency, local government (the Local Government Association and the Local Authority Recycling Advisory Committee), the waste management industry (Environmental Services Association and Biffa Waste Services), the waste management professional association (Chartered Institute of Waste Management), an environmental NGO (Green Alliance), and expert consultants involved in the LATS process (Eunomia and Enviros). In general, access to interviewees was more difficult in the UK. Some key staff in DEFRA at the start of the LATS process in the late 1990s had since then retired or changed jobs. Three organisations that were approached declined to be interviewed, due to lack of time or lack of interest and engagement in the LATS instrument (Confederation of British Industry, Community Recycling Network, Friends of the Earth).

Despite some problems with access to interviewees in the UK, the overall number of and selection of interviewees meant that a saturation point in terms of different views and opinions on the respective EPIs could be reached (in qualitative research it has been estimated to be around 15 interviews, see Kvale 1996 p. 105). Especially in combination with the written material, the interview material was satisfactory.

As described above, careful interview preparation, or 'remote familiarisation', with interviewees (see Phillips 1998) is important for elite interviews. The first phase of document study (spring 2004 onwards) meant that the interviews in the second phase of the research (from May 2005 to February 2006) were well-informed, in terms of the technical issues at hand, the economic interests involved, and the wider political context. A brief topic guide for the semi-structured interviews was prepared (see **Appendix III)** (see also Gaskell 2002; Kvale 1996). After letting the interviewee shortly describe his/her organisation, the idea was to ask questions chronologically about the process rather than ask directly about views on the process and instrument preferences. This would enable the interviewee to gradually recall the process and elaborate on more substantive issues they found important, rather than directly demand more abstract analyses of the process (see Lilleker 2003). The topic guide was not strictly followed so as to allow for more unexpected issues to emerge (see Gaskell 2002, p. 40), and it was also adapted to the interviewee's and his/her organisation's involvement in the EPI choice process. The interviews (of which three were telephone interviews) lasted about one to two hours and were tape recorded, with notes taken during the interview.

Regarding *interview conduct*, the literature on qualitative interviews has discussed at length the problem of subjectivity and the existence of bias. Addressing the question whether interviews are invalid sources of evidence since they convey the interviewees' subjective views, Kvale (1996, p. 66) argues that the interview can be seen as an interpreting method for 'letting the object speak'. Essentially, he advocates a more post-structural stance in that the interview should not seek to establish facts but understand experiences and perceptions in a "linguistically constituted and interpersonally negotiated social world". Indeed, the purpose of the interviews in this study was to capture subjective views of which EPIs were preferable, their rationalization and how an EPI choice process should be conducted (cf. Gaskell 2002 p. 44; Fielding and Thomas 2001 p. 126). Recognising the value of interviews as a data collection method, however, it is crucial to minimise interviewee and interviewer bias and maximise the information value as much as possible. For example, the interviewer may not understand case-specific connotations, interviewees may omit important details, and questions may be leading. These potential biases were avoided as far as possible by careful preparation and understanding of the issues, asking questions in an open fashion, using probing questions, and using a non-technical language (see Gaskell 2002 p. 44; Fielding and Thomas 2001 p. 128; Kvale 1996 pp. 148-149). Furthermore, interview responses could be compared with the written material.

There were two ethical issues relating to the interviews, namely ensuring that the interviewee was informed about the purpose of the interview and confidentiality of responses (see Kvale 1996, p. 114). Each interviewee was briefed about the research project before the interview, either via email or at the beginning of the interview. All interviewees were offered anonymity and since some required this, the decision was made to anonymise all quotes.

The third and final mode of data collection, *direct observation*, was used only for the Swedish case. It was of secondary importance only and did not therefore cause major asymmetries in terms of data availability between the two cases. Permission to attend the meetings of the 2003 inquiry into the incineration tax was granted in the spring of 2004 lasting until April 2005[36]. Normally, the meetings of commissions of inquiry in Sweden are closed for the public. Notes were taken in these meetings of the members' statements and viewpoints.

The analysis of documents, interview transcripts and direct observation notes took place through coding, or conceptual ordering (Strauss and Corbin 1998, p. 73). Coding was first made based on the key elements of procedural rationality identified in **Table 15** above. In parallel, coding was made on the substantive issues involved, for example the actor's position on incineration as a waste management method, the desirability of waste imports, and, most importantly, EPI preferences. A second coding was then made based on the alternative factors explaining EPI choice summarised in **Table 16** above.

To conclude, triangulation of data sources was achieved to some extent by including both documents and interviews in the body of evidence. In an ideal situation, direct observation should have been possible also in the case of the LATS in England.

[36] Observations were made at meetings held on 15 June 2004, 7-8 October 2004, 19 November 2004 and 11 April 2005.

However, this was not feasible given that the process had nearly come to an end, there was no similar forum like the Swedish commission of inquiry, and access to policy-maker meetings as an external observer would likely have been more difficult to secure.

4.4 Concluding remarks

This chapter has described the analytical frameworks and methodology of this thesis, in relation to each of the two research questions. As with all research, the results of this study critically hinge on the research design and methods used. Having presented the methodology above, this concluding section will critically reflect upon three aspects of the research process; the construction of the analytical framework, data quality, and the generalisability of the results. Furthermore, comments will be made on how future studies in this field could be designed.

First, the comprehensive literature review on EPI choice process undertaken for this study was helpful for understanding both basic perspectives on the policy process and more detailed propositions regarding key determinants of instrument choice. The rich palette of existing theories then had to inform the construction of an analytical framework for this study. In the end, frameworks were developed both for studying the procedural rationality of the process and the influence of macro, meso, and micro-level political and institutional factors. An advantage of such comprehensive analytical frameworks is that theoretical triangulation can be applied to the cases (see Peters 1998 p. 146; Allison and Zelikow 1999) and contrasting explanations are proposed to the reader, who can assess their validity him/herself based on the presented evidence. A disadvantage is that more specific propositions cannot be tried and tested in detail. For future studies, results from this comprehensive study could be used to limit the analysis to more specific proposition and a narrower set of variables.

Second, there were some problems with ensuring high data quality within the scope of this study. Regarding the study of the EPI mix (for waste policy and environmental policy generally), the main problem was that the general lack of comparable and complete data meant that nominal measurement only was possible. Building on this and other studies in future research, however, significant time and effort can be saved from basic data collection and spent on more sophisticated analysis, such as examining the relative significance of EPIs (e.g. in terms of environmental effect, cost, perceived importance among stakeholders). Furthermore, the lack of data also meant that some EPI categories were better covered in the analysis of the overall EPI mix, namely economic instruments and VAs. Again, future studies are needed to achieve better balance in the coverage. Future studies could also consider instrument interlinkages and negative EPI choice (i.e. those instruments that were considered but never adopted or old instruments that are removed) to a greater extent. This would provide a more complete understanding of the evolution of the EPI mix. Regarding the EPI choice process case studies, ensuring data quality was not as problematic, except for some asymmetry in the access to data in Sweden and England.

Finally, a critical question is to what extent the results emerging from this methodology are generalisable? There are several levels of generalisation which are relevant here, one of which is the *relevance of municipal waste findings to other environmental policy fields*. As for the mix study, the overall EPI mix is analysed thus eliminating the need for such generalisation. As for the process study, some problem characteristics were discussed above (section 4.2.2) which suggested that instruments choice should be less limited in waste policy than in other policy areas. Thus, generalisation to overall environmental policy overall does not seem excessively difficult. However, in future studies it would be valuable to examine and compare EPI choice in two or more environmental policy fields. In this thesis, I considered including organic farming as a comparison to municipal waste but the time constraints did not permit this extension.

Second, the question is what generalisation can be made *from Sweden and the UK to European policy-making more broadly*. The reasons for choosing these countries were explained above, one of which was familiarity with their environmental policy-making process. A risk is that the researcher in such cases applies a national lens to the data and brings a set of values stemming from his/her political and intellectual culture (Peters 1998 p. 6, 154; see also Rose 1991 and Sartori 1991). This risk was minimised by developing a transparent analytical framework that can be replicated in other country cases. Indeed, future studies examining EPI choice in other countries would be valuable, for example studies of Southern European countries who belong to the laggards in the municipal waste field and the Netherlands and Belgium who belong to the leaders. Furthermore, triangulating this comparative, case-oriented research with more global statistical analyses of some form (see Peters 1998, p. 21) would strengthen the validity of the findings.

Finally, what generalisation can be made from the case EPI choice processes, i.e. the landfill allowance trading scheme in England and incineration tax in Sweden? As described above, more processes could have been studied but at the expense of capturing detail and context of the processes. However, it was also described above that future studies could examine the processes ending up in other instrument choices than economic instruments, in order to understand if there are any systematic differences in process leading up to different EPI choices.

To conclude, it was stated above that the purpose of this research was not rigorous scienctific generalisation regarding the nature of EPI choice. In the qualitative and case study literature, the term of *analytical generalisation* is often used instead (see Yin 1994 p. 31). Kvale (1996, p. 233) defines this form of generalisation as involving "a reasoned judgment about the extent to which the findings from one study can be used as a guide to what might occur in another situation". Arguably, the basis for making such a judgment is two-fold. First, transparency in the procedure of the research is commonly seen as a quality criteria in qualitative research (Gaskell and Bauer 2002 Flick 1998). The aim of this chapter was to ensure transparency in this way. Second, the 'assertational logic' is the main source of validation; "by specifying the supporting evidence and making the arguments explicit, the researcher can allow readers to judge the soundness of the generalisation claim" (Kvale 1996, p. 233). Providing this kind of validation is the task for the empirical chapters to follow.

Chapter 5

The Waste Policy Frameworks in Sweden and England

5.1 Introduction

Before examining instrument choices made over the last decade in the field of municipal waste in chapter 6, the waste policy context in Sweden and England needs to be understood. How was the municipal waste problem perceived? What were the main strategies and policy objectives to be achieved? The purpose of this chapter is to describe the overarching waste policy frameworks, within which policy instruments have been chosen – both the complete instrument mixes and the two case study instruments (the landfill allowance trading scheme in England and the tax on waste incineration in Sweden). First, the nature of the municipal waste problem and its evolution will be compared in the two countries. Then, the traditional policy approaches will be described, followed by a discussion of how new EC waste policy increasingly influenced national policies. In particular, this section will examine how the two countries have responded to the waste hierarchy, which became a key principle for EC waste policy in 1989. In the following section, the concrete implications in terms of waste strategies, objectives and targets at the national level are reviewed. The conclusion will clarify the main implications for the choice of policy instruments.

5.2 The history of the municipal waste problem

Before reviewing the policy frameworks, there is a need to get a better understanding of the character of the municipal waste problem in each country in quantitative terms. Note that there is a problem with lack of reliable and internationally comparable waste data, due to both divergent definitions of various waste streams[37] and poor and infrequent

[37] For example, the UK previously recorded data on household waste instead of municipal waste like many other European countries. Estimates of municipal waste quantities have therefore been made by the

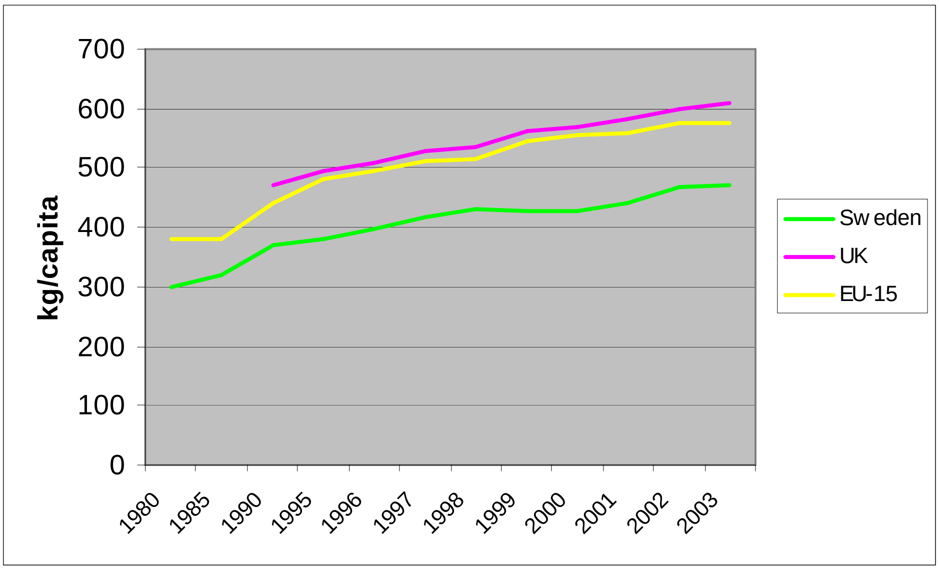
data collection (see EEA 2003 p. 151; Gray 1997 p. 70). In the last decade, international data collection by the OECD and Eurostat has improved, although many figures are estimates only (see Eurostat 2005; OECD 2004a). The voluntary system has now been replaced by an EU waste statistics regulation adopted in 2002 (Haigh 2002, p. 3). Note that in the statistics reported below, Eurostat's data refer to the UK as a whole while DEFRA's data are disaggregated and available for England only. The analysis below will first consider waste generation, then waste collection briefly, and finally the methods of waste treatment; landfill, incineration, and recycling.

Municipal waste makes up about 14% of total waste arisings in weight terms in Western European countries (EEA 2003, p. 153). In Sweden, however, it has been estimated to be significantly lower, around 5% (Eurostat 2005, p. 3), due to larger waste quantities generated by heavy industry (steel, forestry, etc). A significant part of the general waste problem is thus related to industrial and commercial waste, and it was argued by several interviewees in both countries that the strong policy focus on municipal and household waste is misplaced. The municipal waste stream – which is the focus of this thesis, however – consists of about 87% household waste in the UK, with the rest coming from commercial and industrial sites, public spaces, and the clearance of illegal disposal (DEFRA 2005d p. 11; OECD 2002 p. 90).

Increasing municipal waste *generation* is a trend that neither country has managed to curb during the study period of this thesis (1995-2005). In 1995, the total annual municipal waste generation in Sweden was 3.3 million tonnes and in the UK 29 million tonnes (Eurostat 2005, pp. 37-38). The UK thus generated about nine times more municipal waste than Sweden, and it produced 16% of EU-15 total municipal waste (179 million tonnes). Measuring waste generation in *per capita* terms also reveals a significant gap between the two countries. In 1995, Sweden generated 380 kg per capita, while the UK generated 496 kg per capita (see **Figure 4**). The UK exceeded the then EU-15 average (482 kg per capita) and was thus at the outset of the study period one of the worse performers in Europe. Countries performing worse than the UK were Luxembourg, Denmark, the Netherlands, Germany and Ireland. The only country reportedly performing better than Sweden, on the other hand, was Greece.

statisticians. The Eurostat definitions of municipal and household waste are used in this study (see footnote 5, section 1.5) (Eurostat 2005, pp. 33-34).

Figure 4. Municipal waste generation, 1980-2003, kg per capita



Note: The time series is partly discontinuous. Data for 1980-1990 are taken from OECD (2004) and data for 1995-2003 are taken from Eurostat (2005).
Sources: OECD (2004a, table 2A); Eurostat (2005, table A3.1).

By 2003 (the latest year for which comparable statistics are available), municipal waste generation had risen to 471 kg per capita in Sweden, 610 kg per capita in the UK, and 577 kg per capita as an average in EU-15 (Eurostat 2005, pp. 37-38) (see **Figure 4**). These figures equal an increase over the eight-year period of 24%, 23%, and 20% respectively. Thus, while the UK had a more difficult problem in absolute terms, the relative growth of the municipal waste stream has been slightly higher in Sweden. Both countries also belong to the worse performers on this indicator (percentage growth in 1995-2003) compared to the EU-15 average.

The European data for 2003 show that the same countries performing worse on *per capita* generation than the UK in 1995 (see above) did so also in 2003, except for the Netherlands. Sweden, on the other hand, had slipped down the league table, being outperformed by Greece, Belgium, Finland and Portugal. Among the EU member states included in the data, only one managed to actually reduce its annual municipal waste generation, Belgium (-2%). The Netherlands and Finland managed to limit the growth rate to around 9%. Ireland, Greece and Austria, meanwhile, increased their generation by around 40%. Thus, growing waste generation is a significant problem both in Sweden and the UK in comparative terms, although the UK is considerably worse off in absolute terms. Since 2003, the national datasets suggest that generation continues to increase at a stable pace in the UK (DEFRA 2005d, p. 7) and more rapidly in Sweden (RVF 2006c, p. 7).

Ineffective policy instruments could explain part of this trend and will be duly discussed in chapter 6. Another established cause of increasing waste generation is economic growth. It has been found that Swedish waste generation (including industrial waste) probably has been proportionate to GDP growth over time and that it co-varies with the business cycle (Sundberg 2004, p. 21). In the UK, the increase in municipal waste generation has also been closely related to GDP growth (OECD 2002, p. 86). Thus, no decoupling was achieved in either country.

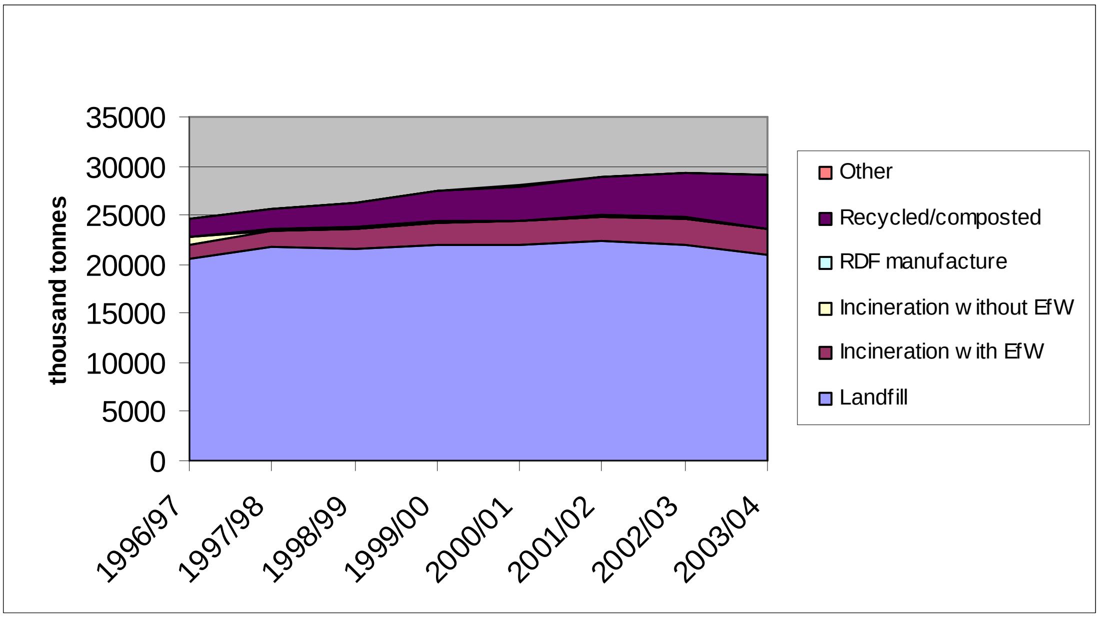
As for the *collection* of municipal waste, the majority is collected through traditional means and about a fifth through separate collection of waste fractions in the UK (Eurostat 2005, p. 83). Separate collection has more than doubled since 1996. No comparable data are available for Sweden, but a 2006 public survey showed that 93% of Swedes separate their household waste at source (Naturvårdsverket 2006b). The level of source separation is directly related to the feasibility and quality of recycling and composting.

The use of the three main *management and disposal* methods for municipal waste – *landfill*, *incineration* (with or without energy recovery [EfW, Energy from Waste]), and *recycling* (material recycling and biological treatment)[38] – is very different in the two countries. DEFRA's Municipal Waste Management Survey 2005 showed that in 1996, landfill was the dominant method and managed 84% of English municipal waste (see **Figure 5**) (DEFRA 2005d). From 1996 to 2003, the proportion of municipal waste sent to recycling and composting increased from 7% to 19%, and recycling and composting capacity more than doubled (215%). This also meant that the proportion of municipal waste sent to landfill decreased to 72% in 2003. However, the reduced proportion of

[38] *Landfill* is a waste disposal method, defined as "deposit of waste into or onto land, including specially engineered landfill, and temporary storage of over one year on permanent sites" (Eurostat 2005, p. 33). *Incineration* is a waste treatment method, which combusts waste. Most modern incineration facilities result in energy recovery (heating and/or electricity). *Recycling* is a waste treatment method, defined as "any reprocessing of material in a production process that diverts it from the waste stream, except reuse as fuel" (p. 34). The Eurostat definition of recycling, and the Swedish national waste dataset, thus distinguish between material recycling and biological treatment/composting (which is seen as recovery of nutrients). The English waste dataset include recycling and composting in the same treatment category.

landfill in the English waste management system is partly a result of the high increase in waste generation. In absolute terms, the amount sent to landfill actually increased by 300,000 tonnes in 1996-2003. Incineration was kept at a constant level in relative terms throughout the period, but in absolute terms it nearly doubled (from 1.4 million tonnes to 2.6 million tonnes). Thus, despite increases in recycling capacity, the landfill problem was not solved during the study period. The Landfill Allowance Trading Scheme (LATS) is one of the instruments developed to address this problem.

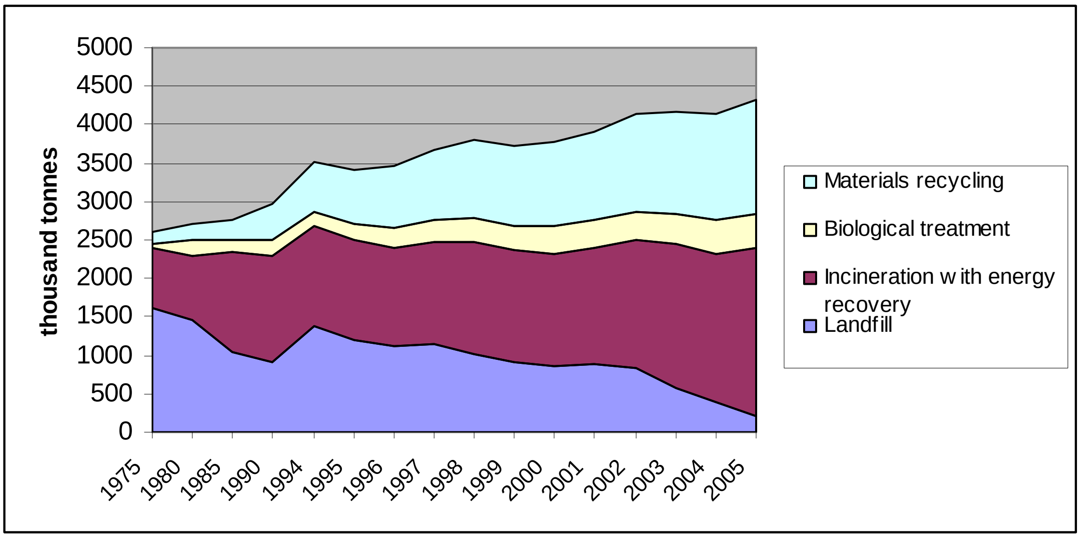
Figure 5. Management of municipal waste in England, 1996/97 - 2003/04, thousand tonnes



Note: Refuse Derived Fuels (RDF) manufacture is defined as "a process whereby municipal waste is compressed into pellets which are then used as a solid fueld supplement in a power station " (p. 79). Less than 30,000 tonnes were reported in each of the categories 'Other', 'RDF manufacture', and 'Incineration without EfW' in 2003, which explains why they are not visible in the diagram.
Source: DEFRA (2005d, Annex A, Table 3).

More historical data is available from the trade association Swedish Waste Management (SWM) on Swedish municipal waste management (see **Figure 6**) (RVF 2006c, RVF 2006b). Back in 1975 more than half of all municipal waste was sent to landfill, and incineration was the other main management method. By 1995, incineration had surpassed landfill as the main method, with about 39% of municipal waste being incinerated compared to 35% being landfilled. Meanwhile, material recycling had expanded significantly by 1995 (to 20%), as well as biological treatment to a lesser extent (to 6%). Over the following decade, these trends continued, with landfill taking only 5% of municipal waste in 2005. The majority of the waste previously landfilled, plus the extra waste arising from increased generation, was in 2005 sent to material recycling (34%) and biological treatment (10%). The capacity of each of these management methods thus more than doubled between 1995 and 2005. The use of incineration also grew by a significant 66% over this ten-year period. The tax on waste incineration studied in chapter 8 was adopted to slow down the expansion of incineration and stimulate the use of material recycling and biological treatment.

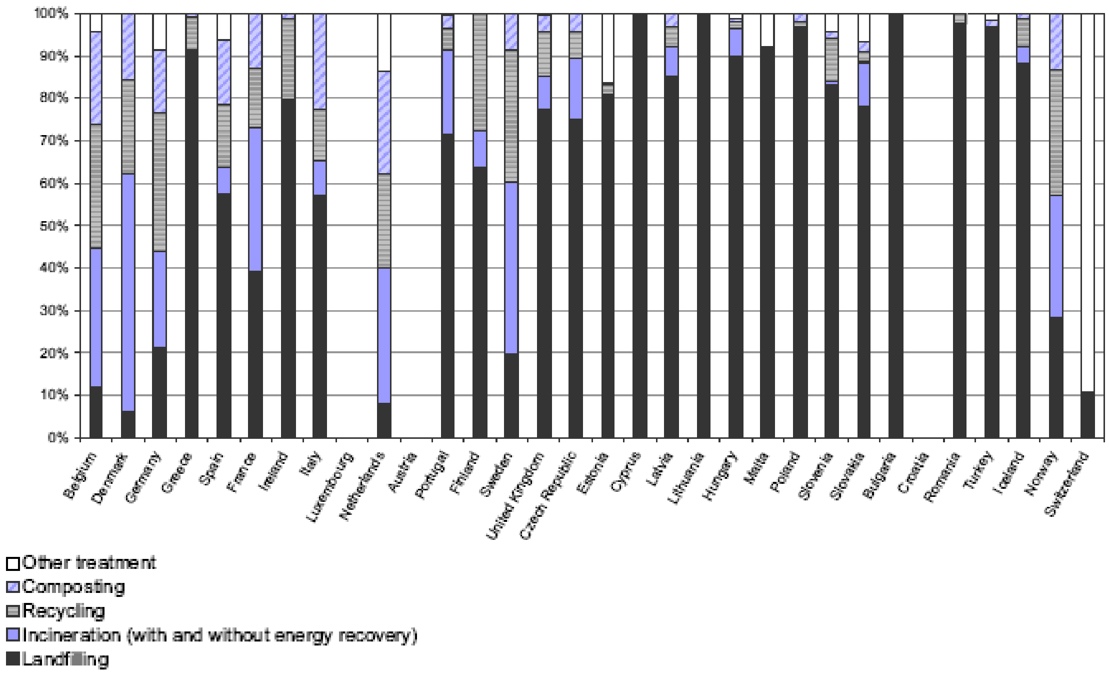
Figure 6. Management of household waste in Sweden, 1975-2005, thousand tonnes



Note: The timeline is discontinuous.
Source: RVF (2006c); RVF (2006b).

England has thus had a historically high reliance on landfill, while Sweden has developed a more balanced mix of management methods and nearly phased out the use of landfill. How do these two countries compare with the EU-15 aggregate levels? Similar to data on waste generation, data on management methods are poor. However, Eurostat (2005) has estimated that the amount of municipal waste landfilled in the EU-15 decreased from around 109 million tonnes in 1995 to 98 million tonnes in 2003 (p. 40). This represents a decrease from 61% in 1995 to 45% in 2003 of total EU-15 municipal waste. However, since total waste generation increased by 20%, the actual decrease in the use of landfill in 1995-2003 was only around 10%. As for incineration, it increased from around 30 million tonnes in 1995 to 41 million tonnes in 2003, i.e. by 37% (p. 42). Given the high increase in waste generation, though, it meant that its relative proportion of municipal waste management only increased from 17% to 19%. In comparison with these aggregated EU-15 statistics, Sweden has thus reduced the use of landfill and expanded the use of incineration much more quickly. England, on the other hand, has had a more stagnant waste management system, although the increase in material recycling and composting has been quicker. The 2002 data in **Figure 7** show how Sweden belongs to the group of countries with low landfill, high incineration and relatively high recycling and composting (together with the Netherlands, Belgium, Denmark and Germany), while UK belongs to the group of countries with a high proportion of landfill.

Figure 7. Use of municipal waste management methods in European countries, 2002, per cent



Source: Eurostat (2005, p. 20, figure 4.10).

In summary, both countries have problems with rapidly growing waste generation, compared with the rest of Europe (although there are worse performers). In particular, Sweden has lost its mid-1990s profile as a low-generation country, as several other countries have curbed their growth trends in more effective ways. In terms of the composition of the waste management system, Sweden has advanced considerably faster and higher up the waste hierarchy than England. Although recycling and composting increased markedly since 1996, England has not yet reduced its dependency on landfill in more than marginal ways.

5.3 Traditional waste policy approaches

The waste management situation in each country in the mid-1990s can, at least partly, be explained by the traditional waste policy approaches. The history of Swedish municipal waste policy seems to be characterised by an early awareness of the need to recycle and prevent waste, but with limited policy action. Waste issues first became integrated in environmental policy in the 1970s (Naturvårdsverket 2002 p. 23; Naturvårdsverket 2005 p. 72). From 1969, the environmental protection law required waste management facilities to be licensed. The 1975 waste bill introduced the principle of producer responsibility, i.e. that producers should be responsible for minimising and managing the waste their products give rise to, although without any concrete instruments to implement it. There was also an emphasis on using waste as a resource, with investment grants available for material recycling technologies. However, these early initiatives failed due to a lack of markets for recyclates. Incineration, meanwhile, had been expanding since the 1960s, until a moratorium was announced in 1985 as a consequence of the public debate on dioxins. The moratorium was abolished one year later when research findings confirmed that health risks from dioxins were tolerably low, and incinerators were more strongly regulated. Incineration has since then not been an issue of particular public salience.

The 1993 waste policy bill provided fresh impetus for Swedish waste management (Regeringens proposition 1993). It stated that the waste problem had to be prevented rather than managed to a larger extent. The key concept was the ‘eco-cycle’ principle (*kretsloppsprincipen*), which meant that everything extracted from nature should be used, reused, recycled or disposed of with the highest resource efficiency and with the least environmental harm possible. It led to the introduction of producer responsibility schemes and promotion of economic instruments for curbing waste generation (Naturvårdsverket 2002 p. 23; Naturvårdsverket 2005 p. 72).

In contrast, the high dependency on landfill in the UK was built up over a long course of time and not really viewed as a major problem until the 1990s (Davoudi 2000). Cheap landfill at the outskirts of the urban areas had replaced incineration as the dominant management method in the 1930s. The economic disadvantage of incineration coupled with fears of air pollution ensured that the use of incineration in the UK became one of the lowest in developed countries. Some recycling initiatives were introduced in the post-war affluence period and the Control of Pollution Act in 1974 required local authorities to examine ways of promoting recycling. However, these initiatives remained marginal and economically disadvantageous compared with landfill when privatisation processes in the 1980s introduced market mechanisms into local authorities’ arrangements for collection and disposal of waste. According to Davoudi, this "further curbed the discourses of waste management to those dominated by economic efficiency" (p. 169). Porter (1998, p. 197) argues that part of the reason for this inertia was the reactive style and consensual relationship between those involved in waste management, who mainly strived to ensure technically effective management. There was close and closed consultation with local affected interests, rather than a high politics approach to the waste problem.

As a consequence of the cheap availability of landfill and lack of central government policy, the number of landfill sites mushroomed in the early 1980s. However, in the late 1980s the 'alarming situation' of contamination and environmental risk led to a shift in the policy agenda away from total reliance on landfill towards a variety of management options (Davoudi 2000, p. 167) and there was a rapid rise in political salience (Bulkeley, Watson et al. 2005). Waste disposal companies began to build up their supply of landbanks for disposal facilities in an unprecedented way in the early 1990s. However, a smaller number but much larger sites meant that both environmental and political pressures became more concentrated (Davoudi 2000, p. 175).

One reason for this increasing salience in the UK was the emergence of more EC waste policy and legislation in the late 1980s (Davoudi 2000 p. 212; OECD 2002 p. 87). In the early 1990s, Sweden also started adapting its waste policy to EC legislation, as a preparation for the 1995 membership in the EC (Naturvårdsverket 2005, p. 73). In contrast to the UK, Sweden had by that time marketed itself as a pioneer and exporter of waste management technologies, a development which had been at least partly driven by regulatory and policy pressure (see Ministry for Foreign Affairs and Swedish Environmental Protection Agency 1999).

The main policy principle to emerge from the EC in the waste field was of course the *waste management hierarchy* (see **Figure 1** in chapter 1), introduced in the 1989 waste strategy (Commission of the European Communities 1989). The hierarchy was welcomed in the 1993 waste policy bill in Sweden (Regeringens proposition 1993). It also provided a strong argument for critics of the UK waste policy approach, which had been seen as unstrategic, incoherent and too landfill-focused (Anon. 1995b).

To summarise, before the growing influence of EU waste policy and legislation in the 1990s, the traditional policy approaches differed considerably between the two countries. According to Davoudi (2000), the UK had for a long time an approach of 'filling holes in the ground'. The gradual shift beginning in the 1990s involved more active development of alternative methods to landfill and a stronger emphasis on consultation with a wider range of stakeholders (see also Gray 1997 p. 70; Read 1999). In Sweden, meanwhile, policy attention to alternative management methods to landfill was stronger and earlier. The concept of the 'eco-cycle' became central in the early 1990s and the focus was gradually shifted upstream in the waste chain, on products and the responsibility of producers (Naturvårdsverket 2002). In Sweden, the waste hierarchy thus reinforced the policy path already embarked upon, while the UK was initially more reactive.

5.4 The influence of EU waste policy

EU waste policy thus increasingly shaped national waste policy from the 1980s onwards. Before the mid-1970s, waste was largely regarded as a local issue for member states (Haigh 2002). In 1975, the Waste Framework Directive was introduced, with daughter directives on waste oils, PCBs and toxic waste. Waste was also a theme in the Second Environmental Action Programme (EAP) (1977-1981). In the 1980s, further legislation was adopted, regarding the transfrontier shipment of toxic waste, sewage sludge and beverage containers (Porter 1998, p. 202).

No longer merely justified in terms of internal market concerns, the first comprehensive waste strategy was issued in 1989 (see **Table 17**). As mentioned above, it established the waste management hierarchy. The waste hierarchy was originally formulated by the OECD in 1976 (HoL Select Committee on European Communities 1998b). It means that, primarily, generation of waste shall be prevented. Waste that is nonetheless generated shall be reused or recycled, through materials recycling, biological treatment or incineration with energy recovery. The strategy avoided ranking these recycling options. As a last resort, waste shall be safely disposed of in controlled landfills (Commission of the European Communities 1993, pp. 7-18). Compared to traditional waste regulation, which focused on the safe and efficient operation of different waste management methods, the waste management hierarchy promoted a 'greener' approach by aiming to reduce the volume of waste.

Table 17. Key EC/EU waste strategies and directives in the 1990s and 2000s

EU waste strategies
1989 – Community Strategy for Waste Management (Commission of the European Communities 1989)
1996 – The Strategy was revised in 1996 (Commission of the European Communities 1996b)
2005 – Thematic strategy on the prevention and recycling of waste (Commission of the European Communities 2005c)
Environmental Action Programmes
1993 – The Fifth EAP (1992-2000) highlighted waste as a key environmental policy issue and emphasised waste prevention. A target was set to stabilise the generation of municipal waste at 1985 levels (300 kg per capita per year) by 2000 (Commission of the European Communities 1993)
2001 – The Sixth EAP (2001-2010) linked waste to resource efficiency and material flows, but set no target for municipal waste generation (Commission of the European Communities 2001a)
Major waste-related EC Directives and Regulations
The key waste-related EC Directives adopted in the 1990s include:
on *horizontal waste legislation* –Waste Framework Directive (75/442/EEC, revised 1991), Hazardous Waste Directive (91/689/EEC), 1993 Regulation on Waste Shipment (No. 259/93);
on *waste management operations* – IPCC Directive (96/61/EC), Landfill Directive (99/31/EC), Waste Incineration Directive (2000/76/EC);
on *specific waste streams* – Batteries Directive (91/57/EEC), Packaging and Packaging Waste Directive (94/62/EC), Disposal of PCBs and PCTs Directive (96/59/EC), End-Of-Life Vehicles Directive (2000/53/EC), Waste Electrical and Electronic Equipment Directives (2002/95/EC and 2002/96/EC).

Besides the waste hierarchy, another policy principle that emerged from the EC waste strategy was the 'proximity principle' (Haigh 2002, p. 1). It suggests that waste shall be disposed of as close as possible to its source. Furthermore, the self-sufficiency principle suggests that each member state should be capable of dealing with its own waste generated in that state (Porter 1998, p. 204).

The 1989 strategy was reflected in the revised 1991 Waste Framework Directive, which placed an obligation on member states to establish a network of disposal installations with the aim of self-sufficiency (Haigh 2002, p. 1). Much of the EU waste policy work in the 1990s was then driven by the agreement on priority waste streams among key parties (*ibid.*, p. 4). Several new directives were adopted for waste streams such as batteries, packaging waste, end-of-life vehicles, and electrical and electronic equipment waste (see **Table 17**). In the early 1990s, negotiations were also initiated on the Landfill

Directive, which due to high controversy could not be agreed upon until 1999. A key provision in this Directive was the quantified targets to limit landfill of biodegradable municipal waste; namely to reduce the amount of biodegradable waste sent to landfill to 75% of the 1995 level by 2006, 50% by 2009, and 35% by 2016[39]. The Incineration Directive followed one year later, mainly involving stricter emission standards for incinerators and no quantified reduction targets.

In 1996, the Commission revised the 1989 waste strategy (Commission of the European Communities 1996b). The waste hierarchy as a guiding principle was kept intact. However, an important addition was made; "preference should be given, where environmentally sound, to the recovery of material over energy recovery operations" (p. 10). Since material recovery was seen as having a greater positive effect on waste prevention, the Commission introduced this ranking. As for policy instruments, a wide variety of instruments was envisaged (Haigh 2002, p. 5). This strategy was developed within the context of the Fifth EAP (1992-2000), which had set targets to stabilise municipal waste generation in the EU at 1985 levels (300 kg per capita) by 2000 and to recycle or re-use 50% of paper, glass and plastics. Almost all countries eventually failed to reach the generation stabilisation target, by exceeding the 1985 level by as much as 75-100% (EEA 2003, p. 154).

Following the introduction of new waste legislation, waste strategy, and EAP waste targets in the 1990s, the focus gradually shifted to consolidation and implementation of existing policy and legislation (Porter 1998, p. 214). EC waste law was increasingly seen as highly complex, with vague and open-ended objectives and definitions of key terms (Tromans 2001). In 2003, the Commission commissioned a report on implementation of five EC waste directives. It was found that there were still significant problems (Haigh 2002, p. 7).

Improving implementation of existing legislation also became one of the priorities in the 2005 revision of the EU waste strategy (Commission of the European Communities 2005c). The Sixth EAP (2001-2010) had initiated work on six 'thematic strategies', of which one was on the prevention and recycling of waste. Other priorities were identified

[39] See Article 5(2) in the Landfill Directive (99/31/EC).

as the introduction of life-cycle thinking in waste policy, more ambitious waste prevention policies, and further elaboration of recycling policy (e.g. through material-specific approaches rather than producer responsibility, quality standards for recycled materials). The use of economic instruments for waste prevention and reduction of landfill was generally promoted. Importantly, the waste hierarchy played a less central role in the thematic strategy, compared to the 1996 strategy. In the 2003 draft strategy it was stated that “while recognising this general principle, the Commission considers that there is a need for further developing approaches for the determination of best environmental options and for the setting of targets for recycling and recovery of waste..” (Commission of the European Communities 2003, p. 18). In particular, no attempt to rank material recycling and energy recovery was made. The final 2005 strategy, on the other hand, saw the hierarchy as a broader aim, but emphasised that it should not be seen as a ‘hard-and-fast’ rule (Commission of the European Communities 2005c, p. 4). Regarding quantified targets, the idea of waste prevention targets had been rejected already in the draft strategy. However, the final strategy also failed to set any targets for recycling, despite early intentions to do so. Increasingly, there is a move within the EU towards considering waste policy within the context of Integrated Product Policy (IPP) and sustainable production and consumption, and viewing waste as a resource.

In summary, there was an outburst of new waste-related EU legislation in the 1990s, that required the member states to develop new EPIs at the national level. The EU waste strategies and EAPs have generally welcomed the use of new types of EPIs, most enthusiastically economic instruments for waste prevention and for limiting landfill. They have also introduced a couple of quantitative targets to guide policy, although the latest thematic strategy refrained from a target-setting approach. Importantly, though, one set of legally binding and quantified targets for member states has been introduced, that implements the waste management hierarchy in a very concrete way; the reduction of landfill of biodegradable municipal waste. Otherwise, the precise interpretation and realisation of the waste hierarchy has been subject to considerable controversy, especially in the debate on the packaging and packaging waste directive, according to Porter (1998, p. 204). In the next section, the support of the hierarchy from the UK and Sweden specifically will be reviewed.

5.4 Major policy objectives and strategies in Sweden and England

Both countries thus accepted the waste management hierarchy as a general framework, but it was used to a varying extent in national strategies and policy. Both countries also had to implement the series of waste-related EC directives adopted in the 1990s. How did they tackle this challenge? Below is a brief overview of the national development in terms of comprehensive waste policy bills and strategies (see **Table 18**), with the aim to provide an overall picture of the objectives and targets for which EPIs have been developed (discussed in the next chapter).

Table 18. Major waste policy bills and strategies in Sweden and England

England	Sweden
1995 – *Making waste work* strategy for England	**1993 – Waste policy bill *On guidelines for a societal development adapted to eco-cycles***
1998 – *Less waste more value* consultation paper on the waste strategy for England and Wales	**1997 – Waste policy bill *Management of used products in an ecologically sustainable society – a responsibility for all***
1999 – *A way with waste* draft waste strategy for England and Wales	**1999 – Government communication *National Waste Strategy***
2000 – *Waste strategy 2000* for England and Wales	**2001 – Environmental policy bill *National Environmental Objectives***
2002 – *Waste not, want not* strategy review by the Strategy Unit	**2003 – Waste policy bill *A society with non-toxic and resource-efficient eco-cycles***
2004 – *Sustainable waste management* consultation paper	***2005 – Environmental policy bill***
2006 – *A review of England's waste strategy* consultation paper	**2005 – SEPA waste strategy *Strategy for sustainable waste management***

As described above, the 1993 waste policy bill laid the foundation for a more proactive waste policy in Sweden guided by the waste hierarchy, and also led to the introduction of producer responsibility principle (Regeringens proposition 1993). In the ensuing 1997 bill, the emerging EU Landfill Directive was a key issue and was responded to by an announcement of several new policy instruments (landfill bans, a landfill tax and stricter landfill licensing requirements) (Regeringens proposition 1997). The next major waste policy bill came in 2003, but before then the management-by-objectives approach to Swedish environmental policy had been introduced (Regeringens proposition 1998). This new system of environmental objectives included targets for waste management and a 'strategy for resource-efficient and non-toxic eco-cycles'. Also, in the late 1990s, the focus on integrated product policy (IPP) increased, reinforcing the 'upstream' approach to waste management (Naturvårdsverket 2002, p. 25). The 2003 waste policy bill confirmed previous policy and introduced few new initiatives, except for the announcement that an inquiry on a new tax on waste incineration would be started (Regeringens proposition 2003). Finally, in 2005 SEPA issued a national waste strategy, partly as a response to EU requirements[40] (Naturvårdsverket 2005). However, it contained few original proposals and no new targets or objectives.

Drawing out the key aspects of Swedish waste policy from the latest bill suggests that it has not changed significantly compared to the traditional policy approach outlined above. The early support of the waste management hierarchy as a guiding principle has continued; "[it] is a basis for the Swedish government's policy in the waste field" (Regeringens proposition 2003, p. 21). Prevention is seen as key, although a rather weak and qualitative only aspiration is stated to 'not increase' waste generation (p. 16). Regarding the ranking of material recycling and energy recovery, a more ambiguous policy has emerged. After referring to several environmental studies, the bill finally states that "recycling measures are ranked, whereby materials recycling is prioritised over energy recovery when this is after a comprehensive assessment environmentally justified" (p. 21). However, it is also emphasised that incineration with energy recovery is the most suitable method for some waste and that it will be needed in the short- and medium-term to manage increasing waste volumes. Interestingly, a somewhat 'greener' approach to the waste hierarchy was expressed in the preparation of the EU thematic strategy (see above). The Swedish government's submission to this process proposed that, in the EU context, the hierarchy needed to be translated into more prescriptive rules and concrete targets in EU waste legislation (Ministry of Environment 2004). It was argued that "[e]ven with well-defined [environmental performance] standards for the different waste treatment methods, it is of environmental benefit to steer the waste higher up in the hierarchy" (p. 6). Beside the support of the waste hierarchy, another key feature of the Swedish policy approach is the aspiration to be an international forerunner in the waste field; "Sweden has through the years developed a progressive environmental policy and implemented large infrastructural initiatives, which has led to an international advantage in developing technological solutions in the environmental field" (Regeringens proposition 2003, p. 20).

[40] Directive 75/442/EEC requires all member states to produce waste management plans. Twelve EU countries have prepared national plans or strategies, and three other regional ones (EEA 2003, p. 162).

Compared to the Swedish development, the waste strategies issued for England since the mid-1990s represent more of a break with the traditional policy approach which had been criticised for being reactive and unstrategic. The 1995 strategy *Making waste work* came as a response to this criticism (DoE 1995). The strategy established the waste hierarchy as a basis for policy, but did not commit to a strict interpretation with clear ranking of options. It was stated that "a single recovery category for recycling, composting and energy recovery indicates that no one of these should automatically be preferred to any other, as this will depend on the Best Practicable Environmental Option [BPEO] for a particular waste stream" (p. 5). The BPEO[41] principle involves a cross-media environmental assessment undertaken by technical staff, as opposed to a top-down policy guideline. Thus, a more flexible and discretionary approach to the waste hierarchy was promoted.

The Labour government that took office in 1997 started consulting on a revised waste strategy in 1998 (DETR 1998; DETR 1999b). A new, highly important point of departure was the draft EU Landfill Directive and targets for the landfill of biodegradable municipal waste that were being negotiated (see above). In 2000, a new waste strategy for England and Wales was finally issued, with greater emphasis on economic instruments and the use of policy targets (DETR 2000a). It was expected to amount to a 'quiet evolution' of UK waste management practices (Porter 1998, p. 200). However, already in 2002 it was subject to critical scrutiny by the Cabinet Office Strategy Unit. In the Unit's review *Waste not, want not* a range of new policy recommendations were made (Strategy Unit 2002). This eventually led to a new round of consultation, and the publication of a major waste strategy review in 2006 (DEFRA 2006).

Key aspects of the evolving England waste policy during this time period were the continued discretionary interpretation of the waste hierarchy. According to Davoudi (2000, p. 178), the ambiguity of the hierarchy as stated in the 1995 strategy and the elasticity of the BPEO principle meant waste policy implementation varied widely at

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[41] BPEO was defined by the Royal Commission of Environmental Pollution in 1998 as "the outcome of a systematic consultative and decision making procedure which emphasises the protection and conservation of the environment across land, air and water. The BPEO procedure establishes, for a given set of objectives, the option that provides the most benefits or least damage to the environment as a whole, at acceptable cost, in the long term as well as in the short term" (DoE 1995, p. 6).

the local level. The waste hierarchy became further demoted from a policy principle to a conceptual framework in the 1999 draft waste strategy, as it became obvious that the aspirations from the 1995 strategy failed and the Landfill Directive would make quick advancement difficult (DETR 1999b, p. 17). The centrality of the hierarchy further decreased in the 2000 waste strategy (DETR 2000a, p. 42), where it was seen as a sub-principle to the BPEO. In the 2002 Strategy Unit review (Strategy Unit 2002, p. 44) it was collapsed into two hierarchical levels only; disposal as one level and recycling, reuse, and reduction as one level. The 2006 waste strategy consultation document merely referred to it in passing. Also in the EU context, the UK has advocated a softer and more pragmatic interpretation of the hierarchy. In response to the draft EU thematic strategy in 2003, it was stated that waste management must be both environmentally, socially and economically rational; "this means assessing whether a particular course of action is economically efficient, socially acceptable and represents the best practicable environmental option" (DEFRA 2003a, p. 6). In addition, the UK emphasised the need for 'robust evidence' to inform any new policy measures, respect for the subsidiarity principle, realistic timeframes, and prescriptive legislation as a 'last resort' instrument.

So, does the waste hierarchy still hold a decade and a half after its adoption in the EU context? Clearly, UK policy-makers have been more sceptical of it and emphasised that time is needed to deal with the scale of the change the country is facing in diverting waste from landfill. English strategies have also consistently referred to the BPEO principle and importance of using good life-cycle analysis (LCA) and data on local conditions when determining how waste should be managed. Sweden has implemented it in a less sceptical way. However, while Sweden implemented policy changes in line with the waste management hierarchy earlier than the UK, there are still unresolved issues. The domestic debate on the relative significance of environmental impacts of different waste management options and what is economically justified is still vigorous (Ekvall, Finnveden et al. 2004), and conflicting evidence has been presented (see for example Finnveden, Björklund et al. 2005).

It was described above that at the EU level, quantified policy targets were increasingly formulated during the 1990s in the context of the waste hierarchy, although this trend seems to have stopped with the 2005 thematic strategy. Obviously, clear and quantified targets provide clearer reference points for choosing EPIs than more vaguely stated policy intentions, such as the waste hierarchy. The trend of target-setting is very visible in both Sweden and England. Over the 1995-2005 time period, a range of waste policy targets[42] were set in England (see **Table 19**). One of the main targets in the 1995 strategy referred to the *reduction of landfill* (from 70% to 60% by 2005). This target was seen as unambitious by many observers and was then superseded by the Landfill Directive targets applying to biodegradable municipal waste (BMW), which are much more challenging. Another target was set for the *recovery* (i.e. recycling, composting and energy recovery) of municipal waste, at 40% by 2005. Although confirmed by the 2000 waste strategy, it was not met by 2005. Still, the 2006 waste strategy review proposed to increase the targets in later years (DEFRA 2006). Finally, a *material recycling and composting* target for household waste was set in 1995, at 25% by 2000. While it was not met by 2000, this target was met by 2005 and higher targets were subsequently proposed in the 2006 waste strategy review. A target for limiting waste generation was never set in any of the strategies, although the latest strategy review clearly has a stronger focus on waste prevention measures. Overall, despite a lack of progress towards several of these targets, the government has thus continued to use a target-setting approach.

Table 19. Waste policy targets in England

Waste generation

No quantitative for capping waste generation has so far been set (OECD 2002 p. 87, DEFRA 2006).

Waste management

- **Reduction of controlled waste consigned to landfill from 70% to 60% by 2005 - 1995 Making Waste Work**

Municipal waste:

- **40% recovery by 2005 – 1995 Making Waste Work**
- **40% recovery by 2005, 45% by 2010, 67% by 2015 – Waste Strategy 2000**

Household waste:

- **25% recycling or composting by 2000; 1 million tonnes per year to be composted by 2001 – 1995 Making Waste Work**
- **25% recycling and composting by 2005, 30% by 2010, 33% by 2015 – Waste Strategy 2000**
- **Increasing recycling and composting at least two-fold by FY 2003, at least three-fold by FY2005 (from FY 1998 level) – Local Government (Best Value) Performance Indicators and Performance Standards Order 2001**

***Biodegradable municipal waste*:**

- **Reducing amount sent to landfill to 75% of 1995 level by 2010, 50% by 2013, 35% by 2020 – Waste Strategy 2000, in line with the EU Landfill Directive**

Sources: DoE (1995); DETR (2000a); OECD (2002, p. 88).

[42] Note that only those targets applying to municipal waste (or fractions thereof) are included here, and not targets applying to other waste streams (e.g. commercial, industrial, agricultural). Targets applying to specific waste streams such as packaging waste, newspring, glass and paper are also excluded here.

In Sweden, targets were not used to such a large extent before the system of national environmental quality objectives, except for the producer responsibility waste streams[43] (see **Table 20**). Like in the UK and EU, a quantified waste prevention target has not been set in Sweden. The first quantified waste management target was introduced in the 1996 waste bill and specified that *landfill* of all kinds of waste (excluding mining waste) should be reduced by 50% of 1994 levels by 2005. This target was motivated by the Landfill Directive and successfully reached in time. In 2003, non-binding targets for *recycling of food wastes* were introduced, with a hint that they could become binding later on depending on progress. A general *material recycling* target was not set until 2005, when it was specified that at least 50% of household waste should be recycled by 2010.

Table 20. Waste policy targets in Sweden

Waste generation

- **The total amount of generated waste shall not increase (introduced in 1998?)**

Waste management

- **By 2005 the amount of landfilled waste (excluding mining waste) shall have decreased by at least 50% from 1994 levels (introduced in 1996)**
- **By 2010 at least 50% of household waste shall be recycled through materials recycling, including biological treatment (interim target no. five, introduced in 2005)**

***Food waste*:**

- **By 2010 at least 35% of food waste from households, restaurants, catering, and shops shall be recycled through biological treatment (interim target no. 9, introduced in 2003)**

Sources: Regeringens proposition (2001; 2003; 2005a).

Similar to the EU level, there thus seems to have been a general trend of setting quantified policy targets in the 1990s, although the Swedish targets were set a bit later. This development of concretising and quantifying desired outcomes has been called for by for example the OECD (see OECD 2002). At the same time, the new UK approach has also been criticised for setting ambitious targets ”but lacking the political will to introduce the necessary policy instruments needed to achieve them” (Humphrey 2003, p. 315).

[43] Within the producer responsibility schemes, quantified targets have been set for glass, plastics, PET bottles, paper and cardboard, wellpapp, aluminium and beverage cans. They are not included here.

5.5 Conclusion

The purpose of this chapter was to briefly describe the waste policy framework in 1995-2005, under which new policy instruments examined in the following chapters have been developed. The aim was to clarify the key policy strategies, objectives and targets, as well as characterise the municipal waste problem in each country.

With regards to the municipal waste problem, both countries have failed to stabilise waste generation which has increased over the last decade. Although the UK generates about a fifth more than Sweden in per capita terms (610 kg/capita compared to 471 kg/capita in 2003), Sweden has experienced a fast growth since the mid-1990s. The question is what instruments have been devised to curb generation and thus advance towards the highest level in the waste hierarchy, prevention. As for the other waste management options in the hierarchy – landfill, incineration, material recycling, and reuse – it was seen that the UK is heavily reliant on landfill, while Sweden has progressed up the hierarchy to a further extent. In 2003, more than 70% of municipal waste was landfilled in England. Sweden, meanwhile, sent more than 40% of household waste to material recycling and biological treatment in 2005. This raises several questions. What kind of policy instruments, or lack thereof, have failed to reduce the use of landfill in the UK? What kind of instruments have contributed to a more balanced waste management system in Sweden? Furthermore, how do policy-makers respond to the different urgency of the waste problem – have they been more creative and diversified the EPI mix more in England? Or have Swedish policy-makers afforded to experiment with new kinds of EPIs to a larger extent?

Moving from the problem character to strategies for waste, it was seen that historically the UK had been reactive and advocated a ‘filling-holes-in-the-ground’ approach. A comprehensive waste strategy was not prepared until 1995. It initiated a break with the old approach and a series of waste strategies urging the need to reduce landfill were issued in the late 1990s and early 2000s. In Sweden, there was earlier awareness of the need to prevent and recycle waste and less of a need to prepare overarching national strategies to change the waste management system. In both countries, however, the policy development has been closely linked to the EU waste-related strategies and legislation, which also intensified during the study period.

There are three aspects of this policy development that are relevant to consider in conjunction to EPI choice. First, the waste policy bills and strategies both at national and European levels have generally advocated the use of new forms of EPIs to tackle the municipal waste problem. This aspiration, together with the significant number of new EU waste-related directives, suggests that one can expect a higher number of and more diverse EPIs now compared to a decade ago.

Second, it was seen that the waste hierarchy was introduced as a key policy principle in the late 1980s. It received mixed support, generally with a higher degree of scepticism in the UK than in Sweden. Over the last decade, it seems to have lost its central role in both countries and in the EU. However, it is relevant to consider the role it has played in specific EPI choice process, when it needs to be translated from a general principle into hard decisions.

Third, in relation to the promotion of the waste hierarchy, quantified policy targets have been increasingly used. These have referred both to waste generation, rates of recycling, and reduction of landfill. While no new policy targets were set at the EU level in the last waste strategy, the UK seems to strongly favour the target-setting approach, especially in comparison to Sweden. The question is how the existence of quantified and precise policy targets influence the EPI choice process. For example, do they lead to consideration of more alternative EPIs or do they favour EPIs that can provide certainty of outcomes? Have concrete instruments been devised to meet the targets, or are the targets not based on what is achievable? Have targets replaced instruments? These three aspects, as well as the difference in waste problem character in Sweden and England, will be considered in the analysis of actual EPI choices made in the following chapters.

Chapter 6

Policy Instrument Mixes for Municipal Waste in Sweden and England

6.1 Introduction

Having presented the analytical framework in chapter 4 and the waste policy framework in chapter 5, this chapter will analyse the main trends in the waste policy instrument mixes in Sweden and England[44]. To what extent has the rhetoric on a diverse EPI mix and 'best-tool-for-the-job' approaches translated into practice? Have economic instruments, voluntary agreements, and information measures been used to an increasing extent, in parallel with or replacing pre-existing command-and-control regulation? This chapter will first present an overview of EPIs used for environmental policy as a whole in Sweden and the UK. After this study of general trends, a more detailed study of instruments for municipal waste follows. All new instruments adopted in the time period 1995-2005 in Sweden and England will be mapped out and categorised, in order to measure instrument diversity. Then, the coerciveness patterns predicted by Doern and van der Doelen respectively will be compared with the actual patterns. Before the conclusion, the overall effectiveness of municipal waste policy in the two countries will be briefly discussed.

6.2 The overall EPI mixes in Sweden and the UK

How successful have Sweden and the UK been at diversifying the overall EPI mix over the last decade? Previous studies have shown that the use of environmental taxes and charges and voluntary approaches internationally has become more widespread, although there is disagreement whether the uptake has occurred 'stunningly fast' or at 'glacial pace' (see section 2.5). As explained in chapter 4 (section 4.2.3), there were

[44] Part of this chapter has been previously published in Persson (2006).

several methodological and data difficulties that needed to be overcome to provide the overview presented below. A collection of sources had to be used, which often reported on EPI use in inconsistent and incomplete ways and drawing on obsolete information. Nevertheless, the data permit an analysis of the use of command-and-control instruments, economic instruments, and voluntary approaches. A brief commentary is then made on the use of education and information instruments, while the liability and management and planning instrument categories were excluded.

6.2.2 Command-and-control instruments

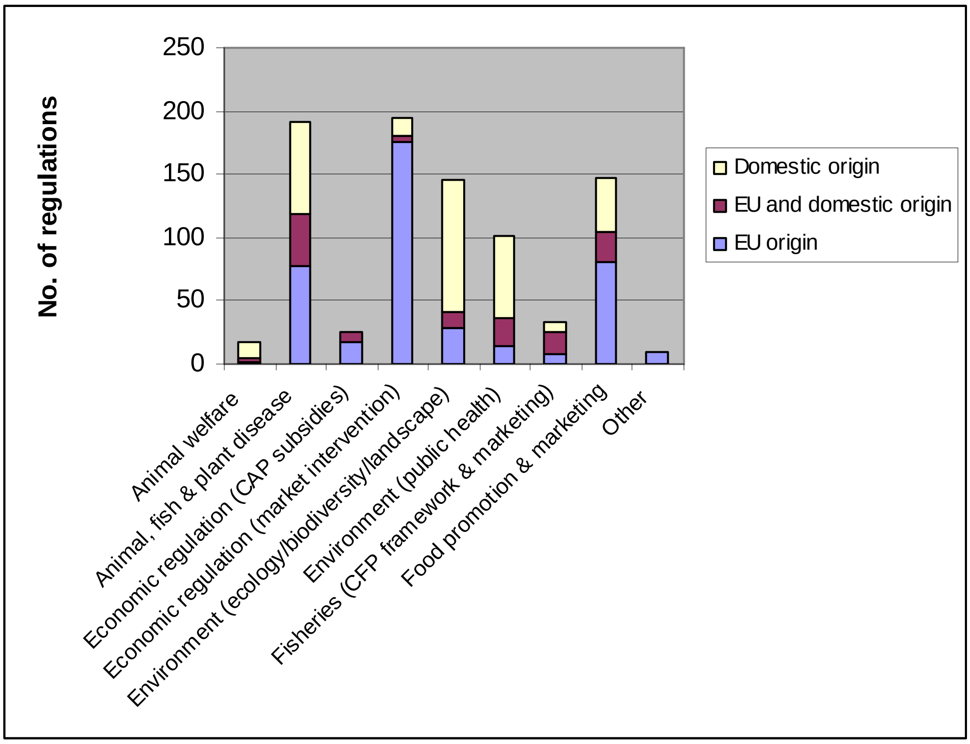
In line with international trends, it seems that NEPIs have not replaced or made redundant the use of traditional command-and-control regulation in either country. In 1996, the OECD concluded that "the use of regulatory instruments remains extremely widespread" in Sweden (OECD 1996, p. 115). Swedish environmental law was reformed in 1999 when the Environmental Code consolidated all environmental regulation into a common framework, and also introduced environmental courts, sanctions, and ambient environmental quality standards (for air quality, fishing waters, and noise). The latter have had a significant impact as new legal instruments (Naturvårdsverket 2004). Overall, the OECD finds Swedish environmental law to be comprehensive and also notes that it sometimes exceed standards stipulated by EU directives (OECD 2004b p. 36; see also Miljömålsrådet 2004 p. 29). In the UK, the regulatory approach has developed from being issue-based to more comprehensive approaches with the 1990 Environmental Protection Act and 1995 Environment Act. The 1996 EU IPPC Directive catalysed a major reform of environmental regulation, towards integrated pollution control (OECD 2002, p. 153). According to Jordan, Wurzel et al. (2003e, p. 194), UK environmental policy is "considerably *more* regulatory in nature and style than it was 30 years ago", mainly due to growing EU environmental legislation.

As described in chapter 4 (section 4.2.3), examining the use of command-and-control instruments is challenging to do in a meaningful and robust way, due to the qualitative nature of such instruments. What is possible, however, is to look for indicators of broader trends. In Sweden SEPA has listed all its regulations[45] applying on 1 January 2006. They totalled 75 regulations (Naturvårdsverket 2006a). Among these, 37 were adopted in 2000-2005, 32 in 1990-1999, and 26 in 1980-1989. This would suggest an increasing, or at least stable, introduction of new regulation into the EPI mix, confirming the general trend that NEPIs are not replacing traditional regulation. However, the limitations of this indicator makes it inconclusive (see comments in section 4.2.3).

In the UK, the DEFRA Regulation Taskforce has presented some findings from the regulation database initiated in 2004 (DEFRA Regulation Taskforce 2004). This database was a work-in-progress at the time and not publicly accessible (see section 4.2.3). It was reported that DEFRA operates 146 instruments related to 'environmental regulation motivated by ecological, biodiversity or landscape concerns' and 101 instruments relating to 'environmental regulation motivated by public health concerns' (see **Figure 8**)[46]. Unfortunately, the database did not report records for earlier years, so the trends over time cannot be discerned. More than half of DEFRA regulations have an EU origin, though. This could suggest that the total body of command-and-control regulation is increasing, given that there is a considerable external input of new regulation. The Taskforce also analysed 215 DEFRA proposals registered in the Cabinet Office's Forward Look database at the time, to examine what regulation was in the pipeline. On this parameter, environmental regulation actually dominated (35%) over the other major areas of regulation (p. 69). Among the environmental regulatory proposals, two-thirds related to EU proposals and UK implementation of international commitments. Thus, the external input to the stock of environmental regulation is significant in the UK.

[45] In Sweden, national laws and regulations are issued at three different political-administrative levels. The parliament issues *laws*. The government issues *ordinances*. The central government authorities issues *regulations*.
[46] In all its fields of regulation, DEFRA operated in total 921 instruments in total, of which 867 were legislative regulations and 54 were non-legislative, such as codes of practice and guidance.

Figure 8. DEFRA regulation categories by origin



Source: DEFRA Regulation Taskforce (2004, p. 65, table 2).

Overall, the Taskforce report found that DEFRA-operated EPIs are 'heavily tilted' towards classic regulation, or command-and-control regulation. It was argued that this was not surprising given the historical use of this type of instrument, relatively recent introduction of the 'alternative EPIs' agenda, and that EU legislation tends to prescribe this form of regulation (p. 43).

The sources reviewed here thus indicate that command-and-control regulation continues to be a key EPI in both Sweden and the UK. The quantitative information provided by the DEFRA Taskforce report and SEPA list of regulations did not permit any conclusions regarding the possible replacement of command-and-control regulations with NEPIs. However, it is notable that the DEFRA Taskforce argues that there is actually a reversed relationship; "orthodox methods will continue to crowd-out advocacy of these alternatives and their use" (p. 43).

6.2.3 Economic instruments

Economic instruments are easier to identify and compare than command-and-control regulation, since they are more discrete entities, have received more attention in policy studies, and can be compared in quantitative terms (rates and revenues). A list of instruments was compiled based mainly on the international OECD/EEA database but also various national sources and is displayed in **Appendix IV**. With consideration to the limitations in data quality addressed in chapter 4 (section 4.2.3), this list suggests there are around 50 economic instruments in use in Sweden and around 30 in the UK (see **Table 21**). These figures refer to instruments currently in use, while some terminated or replaced instruments are also included in the list in the Appendix.

Table 21. Frequency of types of economic instruments in Sweden and the UK

Type of instrument	Sweden	UK
Taxes	11	7
Fees/Charges	11	1
Tradable permit systems	2	5
Deposit-refund systems	3	0
Environmentally motivated subsidies	25	18

There is thus a significantly higher number of economic instruments in the Swedish environmental policy mix than in the UK. Indeed, Sweden has been seen as a pioneer in adopting economic instruments (and environmental taxes and charges in particular) and has "probably more than in any other country" (Ministry of Environment 2003, p. 30). The UK, on the other hand, began using economic instruments later, supposedly due to its long legacy of regulation and general lack of interest in EPIs as such (Jordan, Wurzel et al. 2003e p. 179; Weale 1997). In 1998, Tindale and Hewett (1998, p. 53) argued that there was still more rhetorics than extensive use of them, but in 2002 it was found that there had been significant changes since the shift of government in 1997 (OECD 2002 p. 158, see also Jordan, Wurzel et al. 2003e).

Focusing on specific categories of economic instruments, it is apparent that Sweden has been more keen than the UK to include *environmental taxes and charges* into the instrument mix. It should be emphasised, however, that these data are sensitive to the definition of environmental taxes and charges[47]. Therefore, the frequencies reported

[47] First, taxes and charges are differentiated in that the former are unrequited and the latter are requited. The term levy, encompassing both taxes and charges, has also been used (for example in the UK), to avoid the negative connotations of taxes (EEA 2005, p. 41). Still, some requited levies are defined as

above are subject to some uncertainty. Still, they suggest a clear trend that Sweden has been more willing and interested in taxes and charges as EPIs.

Looking at the *time of adoption* of taxes and charges in each country (see **Appendix IV**), it is apparent that there is indeed a 'pioneer and follower' pattern. While most taxes and charges were introduced in the 1990s in both countries, around a third of the Swedish taxes and charges had been adopted already in the 1970s and 1980s, i.e. before the emphasis on economic instruments in key international environmental policy texts such as Our common future in 1987 and Agenda 21 in 1992. Only three new tax bases were added after 1995 (natural gravel, landfill and waste incineration). In the UK, only traditional energy-related environmental taxes were in use before 1995. The major climate change levy was not introduced until 2001, preceded by the landfill tax in 1996 and aggregates levy in 1996. Interestingly, only a couple of new taxes and charges were introduced in the 2000s in both countries, suggesting a slowing-down or shift in policy-maker interest. However, the number of tax bases is only one side of the equation. The tax rates for several taxes and charges have been gradually raised in the time period studied.

Along with the earlier adoption of environmental taxes and charges as policy instruments, Sweden has also introduced them in a *wider range of environmental policy fields* than the UK. The classification of policy fields in **Appendix IV** shows that the majority of the taxes relate to air pollution and climate change policy objectives in both countries. Sweden has also adopted a couple of taxes for water pollution. In the field of waste policy, Sweden has a broader set of taxes and charges than the UK. Finally, a few different charges related to natural resource management have been introduced in Sweden.

In the UK, the adoption of the new taxes over the study period (1995-2005) has been linked to the Labour government's 1997 statement of intent to use environmental

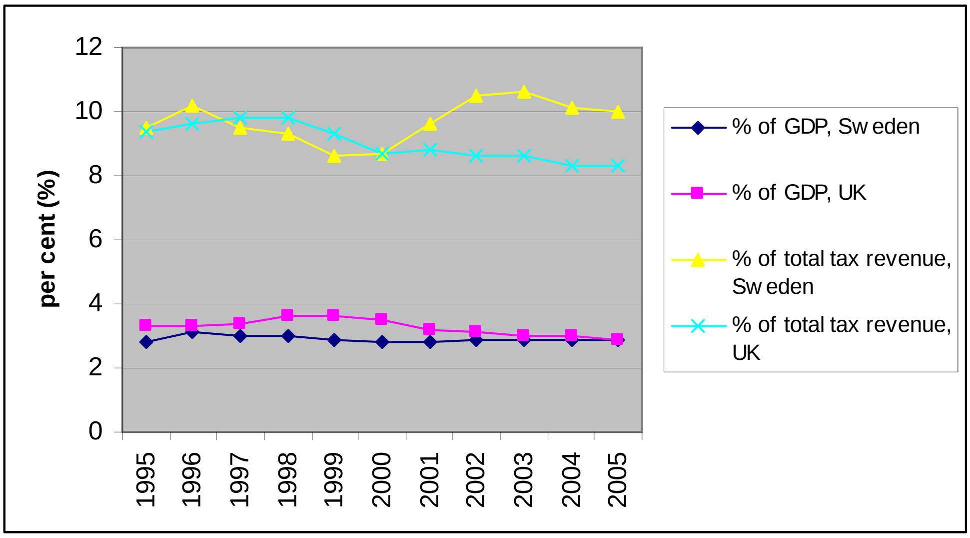
environmental taxes in the UK national environmental accounts (such as the climate change levy and the landfill levy), and thus counted as taxes here. Second, the qualification of some of the taxes as 'environmental' rather than fiscal is questionable. The Eurostat (2001 p. 9, in EEA 2005) definition is that the tax base in question has a 'proven, specific negative impact on the environment'. A recent review of the UK national environmental accounts resulted in one tax included in the data above, the VAT on hydrocarbon duty, (and two others that have been discontinued) being disqualified as an environmental tax (Gazley 2006).

taxation (HM Treasury 2002). In the 2002 follow-up by the Treasury, no new potential tax bases were identified and no firm commitments to taxes as an instrument was made. In Sweden, the major energy tax reform in 1991 paved the way for environmental taxation as an instrument. The process was given new momentum in 2000 when the Green Party negotiated with the governing Social Democratic Party a 'green tax shift' from labour and income taxes to environmental taxes worth 30 billion SEK (3.3 billion EUR) in the period 2001-2010 (Regeringens proposition 2000). Between 2001 and 2006, a shift worth 17 billion SEK had been made (Regeringens proposition 2006, p. 33). However, as shown by the list of taxes in **Appendix IV**, this shift has mainly been implemented through raising tax rates rather than introducing new tax bases.

A more reliable indicator of the use of environmental taxes and charges than their frequency, is the development of *revenues* gained. The official statistics show that the total revenues from the 14 tax bases[48] included in each national dataset were 8.5 billion EUR in Sweden and 52.2 billion EUR in the UK in 2005 (SCB 2005; National Statistics 2006)[49]. Compared to 1995 revenues, the revenues had increased by 55% and 47%, respectively. These increases suggest quite an impressive growth in the use of this new form of EPI, which is of roughly similar magnitude in both countries despite the 'pioneer' and 'follower' pattern. However, eliminating the effects of inflation and general economic growth by relating the revenues to GDP and total tax revenues as in **Figure 9** shows that their significance in the two economies has not increased markedly over time, at least not over the last decade.

[48] The taxes included in these official national environmental accounts have been marked in Appendix IV.
[49] In national currencies, the revenues were 77.6 billion SEK and 35.0 billion GBP respectively.

Figure 9. Revenues from environmental taxes in Sweden and the UK, 1995-2005



Sources: SCB (2005); National Statistics (2006).

As indicated in **Figure 9**, environmental tax revenue in relation to *GDP* has been stable around 3%, with a slight downward trend in the UK. This suggest that environmental taxation has not had a structural effect on the economy. Both Sweden and the UK have a slightly higher ratio than the OECD average of around 2% (OECD 2006, p. 28). A more relevant measure is to compare environmental tax revenue to *total tax revenue*, which was around 10% in Sweden and 8% in the UK in 2005. The OECD average is 6-7% (*ibid*.), suggesting that Sweden and the UK are both significant users of environmental taxation internationally. Although the reasons for the downward trend in the UK over the last decade on this indicator cannot be explored fully here, it is notable that they are consistent with the UK policy to use environmental taxes primarily to change behaviour rather than generate additional state revenues (HM Treasury 2002). The upwards trend from 2000 onwards in Sweden, meanwhile, is consistent with the green tax shift that was initiated at that time. Overall, the key message from **Figure 9** is that after 1995, environmental taxes have become less significant in the UK measured in terms of revenue, while more so in Sweden.

Another measure to understand the role of environmental taxation is the composition of the revenue. The general OECD pattern is that motor fuels and motor vehicles dominate as environmental tax bases, by providing more than 90% of revenues from environmentally related taxes in the OECD as a whole (OECD 2003b). The national environmental accounts of Sweden and the UK for 2005 suggest that this holds true. In Sweden, the revenues from non-energy and non-transport related taxes comprised about 2% of total environmental tax revenue (SCB 2005). In the UK, the same figure was 6% (National Statistics 2006). Thus, non-traditional environmental taxation thus plays a rather minor role in the repertoire of EPIs when measured this way.

Finally, the OECD/EEA database also shows that the *structure of exemptions* related to a particular tax, both in terms of products and sectors of the economy excluded from its scope, is often quite complex in both Sweden and the UK. This suggests that the economic efficiency rationale often has to be balanced with equity concerns. As an alternative to exemptions, the information in the database suggests that refunds and earmarking of tax reveneues are relatively common measures to ensure compromises between efficiency, effectiveness and equity concerns. For example, the Swedish manufacturing industry receives large refunds from the energy and CO2 taxes and the UK climate change levy is refunded to tax payers partly through reduction of national insurance contributions (NICs). Other examples of earmarked levies are the landfill tax in the UK, which is returned through reduction in NICs and credits in exchange of landfill operators' contributions to certain environmental bodies, and the charge on NOx emissions in Sweden, which is returned in proportion to total energy output, hence favouring NOx-efficient facilities.

To summarise the overview of environmental taxes and charges as EPIs, it is clear that Sweden has many more tax bases and charges and that revenues play a larger part in the government budget. However, it has also been found that the Swedish taxes are often too low to cover externalities and to significantly affect behaviour (OECD 2004b, p. 47). The UK taxes and charges, on the other hand, are hypothecated to a larger extent. The introduction of new tax bases was most intense in the 1990s and this trend seems to have slowed down in the 2000s.

In contrast to the slowing-down of new environmental taxes and charges, several new *tradable permit systems* (TPSs) have been introduced in the 2000s. **Table 21** shows that the UK has so far introduced five such systems, compared to Sweden's two (see also **Appendix IV**). The UK has now been called a world leader in designing emissions trading systems, particularly through its pioneering of the greenhouse gas (GHG) emissions trading scheme in 2002 (OECD 2002, p. 180). The first one for recycling of packaging (i.e. the Packaging Recycling Notes) was introduced in 1998, followed by the GHG emissions trading scheme and the tradable renewable energy certificates scheme (i.e. the Renewables Obligation) in 2002. In 2003 the new water act made it possible to trade water abstraction licences in the UK, while the latest TPS instrument added to the mix is the 2006 Landfill Allowances Trading Scheme, which will be reviewed in detail in the case study in chapter 7. Tradable emission permits as an instrument has not been a trademark of the Swedish EPI repertoire. The first one introduced in Sweden was the tradable renewable energy certificates scheme (i.e. the Green Electricity Certificates) in 2003, which works in a similar way to the UK version. The second was the national implementation of the EU GHG emissions trading scheme in 2005. One reason to the earlier and wider uptake of the TPS instrument in the UK could be that the markets in question are larger than the Swedish ones, allowing the TPSs to work more efficiently and at lower transaction costs. Another reason could be that TPSs are often compared to tax designs when considered and it was described above that Sweden has had a more proactive environmental tax policy than the UK.

Deposit-refund systems have not been used at all in the UK, while Sweden has introduced three since the 1970s (see **Table 21**). As policy instruments they are by nature limited to the field of waste management and resource efficiency, and those identified in this exercise will be reviewed further below.

Finally, environment-related *subsidies and grants* are a difficult category of EPIs to compare systematically, since they come in various forms (e.g. one-off investment grants, R&D support, continuous price subsidy). Furthermore, the classification can be complicated by the fact that indirect subsidies could be identified, in the form of tax rebates or exemptions. Recently, there has been an increasing interest in environmentally harmful subsidies, i.e. subsidies which do not have environmental improvement as a primary purpose (see OECD 2003a; EEA 2005 pp. 101-102). For this study, however, only direct and positive environmental subsidies and grants are considered.

Providing a reliable and internationally comparable nominal measure of the number of subsidies and grants is more difficult than for taxes, since they are not yet included in the national environmental accounts as a standard. Nevertheless, the inventory undertaken for this study (see list in **Appendix IV**) suggests that there are around 25 subsidy and grant schemes in Sweden and around 20 in the UK (see **Table 21**). Thus, there does not seem to be a significant difference in nominal terms in the use of this EPI in the UK and Sweden. In both countries, most schemes have been devised for agri-environment policy and natural resources management. Also in the field of energy efficiency and climate change, a few programmes have been launched.

Lacking more precise data, the expenditure on environmental subsidies and protection is a relevant indicator. Official OECD statistics reveal that the public sector expenditure on pollution abatement and control amounted to 0.4% of GDP in the UK and 0.2% in Sweden in 2000 (OECD 2004a, p. 281). While this figure has been stable over time in both countries, public R&D expenditure for control and care of the environment has fluctuated a lot over the last two decades; from about 65 million USD (at 1995 price levels) in 1990 to 15 million USD in 2001 in Sweden, and from 125 million USD in 1990 to 200 million USD in 2000 in the UK (OECD 2004a, p. 283). The figures vary from year to year, making it difficult to discern any trends or patterns.

The national statistics offices in Sweden and the UK have both produced more disaggregated data, although with different methodologies. In the UK, records of subsidies and grants (including capital grants) to industry and households in six different categories exist for the time period 2000-2004 (National Statistics 2005) (see **Table 22**). These data show that expenditure decreased from 661 million GBP in 2000 to 208 million GBP (310 million EUR) in 2004. The time period is too short for observing trends, other than substantial fluctuation from year to year. There is also a lack of description of these statistics, in terms of what they include. Compared to the environmental tax revenue in 2005 – 35 billion GBP – the total expenditure is clearly low, though.

Table 22. Environmental protection expenditure by the UK public sector, 2000-2004

Grants and subsidies, million GBP	2000	2001	2002	2003	2004
Protection of ambient air and climate	201	255	201	155	68
Waste water management	33	165	10	31	71
Waste management	2	9	4	2	2
Protection of biodiversity and landscape	117	120	50	31	31
*Other abatement activities**	220	288	49	2	3
Research and development, education and administration	88	164	60	34	33
TOTAL (million GBP)	**661**	**1,001**	**374**	**255**	**208**

*Includes expenditure on the protection of soil and groundwater, on noise and vibration abatement, on protection against radiation and on other environmental protection activities.
Note: Current and capital grants and subsidies (to industry, households and public corporations) have been aggregated in this table. For disaggregated data, see the original source.
Source: National Statistics (2005).

For Sweden, there is more disaggregated data, reported per instrument, and covering the time period 1993-2004 (OECD 2004b p. 119; SCB 2006). The total environmentally motivated subsidies increased from 0.4 billion SEK in 1993 to 2.6 billion SEK in 1999 (in current prices) (OECD 2004b, p. 119). This meant that it increased from 0.02% of GDP to 0.10%. Almost the whole increase can be attributed to agriculture-related subsidies, which were introduced after the 1995 membership in the EU, as well as grants to agricultural R&D. With the new accounting methodology from 2000 onwards, the total expenditure on environmentally motivated direct subsidies was 6.6 billion SEK in 2000 and 5.8 billion SEK (640 million EUR) in 2004 (SCB 2006) (see **Table 23**). With this methodology the expenditure in relation to GDP is lower again, 0.2%. The largest factor explaining this decrease is the reduction in agricultural subsidies, but there were also a number of new subsidies introduced in this period. Compared to the revenues from environmental taxes – 77.6 billion SEK in 2005 – the expenditure on direct subsidies and grants is higher than in the UK.

Table 23. Total environmentally motivated direct subsidies in Sweden, 2000-2004

Grants and subsidies, million SEK	2000	2001	2002	2003	2004
Natural resource management	5,498	5,349	5,423	4,782	4,606
Energy	1,047	1,239	1,526	1,039	973
GHG emission reductions	12	18	22	43	250
Transport	6	0	0	0	0
TOTAL (million SEK)	**6,563**	**6,606**	**6,971**	**5,865**	**5,829**
% of total subsidies in Sweden	1,5%	1,4%	1,5%	1,3%	1,2%
% of GDP	0,3%	0,3%	0,3%	0,2%	0,2%

Note: Individual subsidy schemes are reported in the original data, but only the five categories are reported here.
Source: SCB (2006).

Overall, in nominal terms there seems to be the same magnitude of subsidies and grants in Sweden and the UK, and they are used mainly for agri-environmental and natural resource management policy. In monetary terms, the aggregate OECD statistics reported that the UK spends more as a proportion of GDP than Sweden (0.4% compared to 0.2%). However, the national datasets – which are based on different methodologies – reported that the UK spent about 310 million EUR on subsidies and grants in 2004, while Sweden spent 640 million EUR. It is thus possible that subsidies and grants as EPIs are more accepted and actively preferred in Sweden than in the UK. Considering the higher use of unrequited environmental taxation in Sweden also, Swedish environmental policy thus seems to be characterise by more redistributive instruments than UK environmental policy.

6.2.3 Education and information

Differences in the use of information measures and regulation between Sweden and the UK are difficult to assess at an overall level and with quantitative measures. The sources used for this review did not explore this EPI to a considerable extent. However, with regards to the use of eco-labels, Jordan, Wurzel et al. (2003e p. 180) state that the UK has in general been much less enthusiastic about them than Sweden and Germany. Sweden was a forerunner in Europe with the ‘Nordic Swan’ ecolabel, and the NGO-led ‘Good Environmental Choice’ label is also widely recognised (OECD 1996, p. 128). In the UK, eco-labels have not been extensively used as a public policy instruments, but rather as private business initiatives (Jordan, Wurzel et al. 2003e, p. 193). Interest from policy-makers did not increase until the late 1980s, and the ideas of a national scheme was then overtaken with the emergence of the EU scheme. The success of this scheme was very low, and DEFRA now encourages the use of EMAS instead. Some businesses focus on larger branding campaigns instead of common ecolabelling schemes.

6.2.4 Voluntary approaches

Finally, voluntary agreements (VAs) do not seem to be a particularly favoured choice of EPI in either country, compared with other European countries such as Germany and the Netherlands (OECD 2002 p. 151; Ministry of Environment 2003 p. 34). According to the list compiled for this study (see **Appendix IV**), there are currently 10 in use in the

UK and 14 in Sweden. However, the data is taken from secondary sources and most likely out-of-date. Still, the trend seems to be that there is roughly a similar number of VAs in both countries and most of them were introduced in the 1990s.

In Sweden, a review by the Swedish EPA in 2000 showed that there were more VAs adopted in the 1990s than previously expected (Naturvårdsverket 2000b). The SEPA identified 17 VAs, but some of these were of such a one-off nature and applying to a specific actor so they should not be regarded as systematic EPIs for this study. In 2004, the OECD's review of Sweden found that the use of VAs had increased somewhat, but they were not yet a key feature of Swedish environmental policy (OECD 2004b, p. 50). Generally, they have been used in preparation of legislation and in combination with other instruments, and have not set particularly ambitious objectives.

Paradoxically, according to Jordan, Wurzel et al. (2003e p. 190), British policy has always been "relatively voluntaristic", but there was still suspicion against more structured VAs in the mid-1990s. Since then, there has been only a cautiously positive approach. Industry has been divided on the merit of negotiated agreements, pointing to risks such as free-riding and impracticalities given the number of trade associations. In total, Jordan, Wurzel et al. (pp. 191-192) have identified around 20 VAs in the UK, around half of which are negotiated agreements and the other ones unilateral commitments or public voluntary schemes. Jordan et al.'s review describes how the first VAs introduced already in the 1950s (in the field of pesticide safety) were 'gentleman's agreements'. The later generation (in fields such as energy efficiency, climate change, ozone depleting substances, washing products) of negotiated agreements often have some official status, but not legal status. The success has been varied, with some being ineffective, criticised for being little more than codified business as usual, and some risking free-rider problems. However, Jordan, Wurzel et al. (p. 195) argue that DEFRA may now be changing its attitude towards VAs, seeing them as particularly relevant in the fields of waste minimisation and resource productivity. They may increasingly be used as tools for setting tentative targets and 'best-practice' benchmarking, potentially evolving into negotiated and binding agreements over time.

6.2.5 Summary of findings

Various measurement problems mean that it is difficult to compare instruments other than in the categories of economic instruments and voluntary approaches. *Command-and-control regulation*, however, has not decreased in either country but rather grown, partly as a result of EU legislation. There are more than twice as many *environmental taxes and charges* in Sweden compared with the UK, but most of them were introduced in the 1990s and this trend seems to have slowed down in the 2000s. Although revenues have increased considerably over the last decade in absolute terms (and mostly due to higher fuel and motor vehicle taxes), the trend in the UK is that they constitute a decreasing proportion of total tax revenues. This suggests that either tax rates are low or the purpose of the taxes was to change behaviour (and thus erode the tax base) rather than to raise revenue. In Sweden, the revenue-raising purpose came to the fore with the 2001 green tax shift programme, which has meant that the proportion of total tax revenue is now increasing. Considering that *tradeable permit systems* have not been a key feature of European environmental policy, it is a significant trend that the UK has adopted four new ones in the 2000s. It is a less preferred instrument in Sweden, especially compared with taxes and charges. Regarding *subsidies and grants* as positive economic instruments, large fluctuations in such expenditure from year to year were seen in both countries. The data for Sweden, which is more disaggregated and spans a longer time period, suggest that there has been a significant increase in the use of this EPI over the last decade, though. Finally, *voluntary approaches* in the form of negotiated agreements is not a particularly favoured EPI in either country, although the UK started using them as early as in the late 1950s. Among the negotiated agreements that do exist, most of them were introduced in the 1990s.

To sum up, the trend is that the overall EPI mixes in both countries have diversified over the last decade. However, most of the NEPIs were introduced in the 1990s, suggesting that the diversification trend has slowed down somewhat in the 2000s. On the whole, Sweden seems to have been more active than the UK in diversifying the overall EPI mix, but more information on the EPI categories not analysed above is needed to draw such a conclusion.

6.3 Types of EPIs for waste policy

So far, this chapter has demonstrated that the quest for new EPIs has been rather successfully translated from rhetoric to reality in both Sweden and the UK. Is this pattern reflected also in the field of municipal waste policy? The remainder of this chapter will focus on diversification of the EPI mix in this particular policy field. Note that it will from now on discuss waste policy instruments applying in England, as opposed to considering the whole of the UK. The results of the complete instrument inventory are graphically presented in **Appendix V** (Table V-A and V-B; grey-shaded cells represent instruments and indicate when introduced) as well as per EPI category in the sections below[50].

6.3.1 Overall diversity

The first striking thing about the evolution of the instrument mixes illustrated in the inventory (see **Appendix V**, Tables V-A and V-B**)** is that the *number of EPIs* has increased significantly in both countries over the time period studied, about a doubling in Sweden (from ten to 22) and a six-fold increase in England (from five to 36). Given the limitations of nominal measurement of EPIs, it is still clear that both countries, and especially England, have chosen to adopt larger and more comprehensive mixes. Rather than relying on a single-instrument approach, the waste problem is now being tackled in a multitude of ways.

Overall diversity, in terms of the *variety of different EPI types* adopted at the national level, has increased in both countries between 1995 and 2005. In addition to command-and-control instruments and management and planning measures, Sweden was an early user of deposit-refund systems (cars, aluminium cans, plastic bottles) and legally based producer responsibility (packaging, newspaper, car tyres). The instrument mix has subsequently been widened, with the introduction of other economic instruments, information measures and a VA. England started with a somewhat narrower range of EPI types, but the last decade has witnessed a burst of new instruments across all EPI categories, coinciding with major waste strategy initiatives. The *Waste Strategy 2000* (DETR 2000a) led to the establishment of the Waste and Resources Action Programme

[50] For sources of information to the tables presented in sections 6.3.2-6.3.7, see source information for the complete tables in Appendix V.

(WRAP) and the 2002 *Waste not want not* strategy review (Strategy Unit 2002) led to the Waste Implementation Programme (WIP). WRAP[51] is a not for profit company started in 2000 by the UK government charged with work on waste minimisation, providing recycling advice for local authorities, and influencing public behaviour through communication programmes. WIP is a programme of policy measures on local authority funding and support, technology development, waste data, and waste minimisation and awareness programmes (several of which are implemented through WRAP) (DEFRA 2005g).

The results seem to support the observation by Jordan, Wurzel et al. (2003a, p. 205) that the UK, together with the Netherlands, have come furthest in Europe with experimenting with mixes, in particular the climate policy area. However, the relative significance (in terms of environmental, economic or political impact or symbolic meaning) of various EPI types or individual instruments cannot be measured simply in nominal terms. Furthermore, diversity as measured here is not difficult to achieve, prompting more detailed analysis of instruments within each EPI category.

6.3.2 Command-and-control instruments

The key command-and-control instruments are displayed in **Table 24** and **Table 25**. *Permitting* of waste management facilities, including specification of emission limits and handling procedures, is a backbone in municipal waste policy in both countries, and the EU Landfill (1999/31/EC) and Waste Incineration (2000/76/EC) Directives led to a revision of regulation in both countries. To meet the target for landfill reduction of biodegradable municipal waste, Sweden introduced *bans* for sorted combustible waste and organic waste in 2002 and 2005 (although with a rather extensive system of exemption licensing). Possibly due to the more challenging situation facing England to meet its landfill targets, *statutory landfill targets* for local authorities rather than bans were introduced in 2005 (through the Landfill Allowance Trading Scheme (LATS), categorised here as a tradable permit system). These have been introduced on top of the *Statutory Performance Standards* for local authorities specifying recycling levels of household waste. Notably, no major coordination exercise for these interrelated sets of statutory targets preceded the LATS introduction (Martin Cox, *pers.comm.*).

[51] See www.wrap.org.uk.

Table 24. Sweden: command-and-control instruments

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
COMMAND-AND-CONTROL INSTRUMENTS											
Licenses/permits,	Waste management site licensing (permitting procedure)			*Amended regulation (SFS 1998:899)*							
							Emissions standards landfill sites *(SFS 2001:512)*				
								Emissions standards waste incineration *(SFS 2002:1060)*			
Prohibition bans				Obligatory batteries collection *(SFS 1997:645)*							
								Ban on landfill of combustible waste *(SFS 2001:512 9 §) (combined with exemption licenses, NFS 2001:117)*			
											Ban on landfill of organic waste *(SFS 2001:512 10 §)*
Statutory targets	(None)										

Table 25. England: command-and-control instruments

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
COMMAND-AND-CONTROL INSTRUMENTS											
Licenses/permits,	Waste Management Licensing Regulations 1994 (1994-)										
								Landfill permits and emission standards (*Landfill Regulations 2002*)			
								Emission standards (*Waste Incineration Regulations 2002*)			
Prohibition bans	(None)										
Statutory targets							Statutory Performance Standards for the recycling of household waste (*Local Government (Best Value) Performance Indicators and Performance Standards Order 2001*)				

Command-and-control instruments still seem to play a very important role in municipal waste policy. The introduction of 'new' EPIs should be seen as an evolution rather than a revolution (cf. Jordan, Wurzel et al. 2003a). A future issue may be how command-and-control instruments will be used as municipal waste policy moves further towards a waste reduction and minimisation focus. Increased use of mandatory material resource efficiency standards for products and even product bans has been raised (OECD 2001a), but no specific instruments to these ends have materialised yet in either country. The UK government has made clear that it prefers prescriptive legislation in the waste policy field as a 'last-resort' EPI only (DEFRA 2003a), while the Swedish government has welcomed more prescriptive instruments at an EU level (Ministry of Environment 2004).

6.3.3 Economic instruments

Among economic instruments, Sweden introduced *charges* for batteries and imported aluminium cans at an early stage (1987 and 1983 respectively), to cover collection and recycling and treatment costs (see **Table 26** and **Table 27**). Both the UK and Sweden have introduced *landfill taxes*. Even though Sweden is a keen user of environmental charges and taxes, the landfill tax was introduced four years later than in the UK. In the UK the total receipt GBP 502 million in 2001/02 (Davies and Doble 2004 p. 77), while in Sweden it was SEK 906 million in 2002 (SCB 2006), i.e. about 0.05% and 0.04% of GDP respectively. Two important differences stand out. First, the tax rate in the UK for non-inert waste has consistently been considerably lower than the Swedish one. In order to accelerate the achievement of the landfill targets, it was announced in 2003 that the UK tax rate should be about GBP 35 (c. EUR 50) per tonne in the medium term (Davies and Doble 2004, p. 77) – thus shifting from an externality valuation rationale to a behaviour change rationale. Second, the revenue of the UK landfill tax is recycled to the waste management industry, through reduced national insurance contributions, tax rebates for contributions to registered environmental bodies (the Landfill Tax Credit Scheme) (see e.g. Martin and Scott 2003), and from 2005 through the Business Resource Efficiency and Waste Programme. The revenues of the Swedish landfill tax, on the other hand, are not recycled but go to the general budget.

Table 26. Sweden: economic instruments

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
ECONOMIC INSTRUMENTS											
Taxes and charges	Charge on imported aluminium cans (1983-2005) *(SFS 1983:847)*										
	Battery charge (1987-) (*SFS 1997:645*)										
						Landfill tax: EUR 27 per tonne *(SFS 1999:673)*		EUR 31 per tonne	EUR 40 per tonne		
Subsidies and grants				Local investment programmes (1998-2002) *(SFS 1998:23)*							
											Investment grant for sorting facilities *(SFS 2004:1387)*
Transferable permit systems	(None)										
Depost-refund systems	Deposit-refund system: cars (1975-) *(SFS 1975:348)*										
	Deposit-refund system: aluminium cans (1982-) *(SFS 1982:349)*									(Incorporated into producer responsibility for packaging in 2003)	
	Deposit-refund system: plastic bottles (1991-) *(SFS 1991:336)*									(Incorporated into producer responsibility for packaging in 2003)	

Table 27. England: economic instruments

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
ECONOMIC INSTRUMENTS											
Taxes and charges		Landfill tax: EUR 4.80 per tonne non-inert waste; EUR 1.40 per tonne inert waste (*Landfill Tax Regulations 1996*)									
Subsidies and grants	Recycling Credits (1992-) (*Environmental Protection Act 1990*)										
		Landfill Tax Credit Scheme (*Landfill Tax Regulations 1996*)									
						Home Composting Scheme (WRAP)					
						Retailer Initiative and Innovation Fund (WRAP)					
						Materials Programme (WRAP): glass, Plastics, Paper, Wood, Organics, Aggregates					
						Recycling Fund (WRAP)					
						eQuip Residual Value Guarantee (WRAP)					
									National Waste Minimisation and Recycling Fund		
										Waste Research Programme (WIP)	
										Demonstrator Programme (WIP)	
										Community Sector Support Programme (WIP)	
											Waste Performance and Efficiency Grant
											Business Resource Efficiency and Waste Programme (BREW)
Tradeable permit systems			Packaging Recovery Notes (implements packaging producer responsibility) (*Producer Responsibility Obligations Regulations 1997; Packaging Regulations 1998*)								
											Landfill Allowance Trading Scheme
Depost-refund systems	(None)										

Note: The annual changes in the landfill tax rate are indicated in the complete EPI table V-B in Appendix V.

By 2005, 15 EU member states had introduced landfill taxes. In about half of these countries, the tax revenue was allocated to the general budget. In the other countries, the landfill charge revenues were used for modernisation and environmental funds, clean-up of contaminated land, municipal budgets, or reduced VAT or national insurance contributions (EEA 2005, p. 64). There was also great variation in the landfill tax/charge rates, from EUR 79 per tonne in the Netherlands to EUR 15 per tonne in Finland (EEA 2003, p. 162).

Although the time period studied here is 1995-2005, it should be noted that a waste *incineration tax* was introduced in Sweden in 2006. This instrument choice process is analysed as a case study in chapter 8. Incineration taxes are also used in Denmark, the Netherlands, Norway, and Flanders, and are related to the landfill taxes in different ways (EEA 2005, p. 63). In the UK, a waste incineration tax has also been considered recently, but was rejected in the 2004 Pre-Budget Report (HM Treasury 2004b, p. 153). It was argued that it would be counterproductive given the need to reduce landfill, and increased recycling capacity alone could not meet this need. Furthermore, it was argued that the negative environmental externalities arising from incineration could not justify a tax. In contrast, the Swedish incineration tax was never justified as an internalisation of external cost, but as an effective measure to influence the relative prices in favour of material recycling (see further discussion in chapter 8).

Waste collection charges, or *user charges*, are commonly seen as an economic instrument for waste policy. However, they are not included here as they are a matter for local government in both countries. It can be noted, however, that the UK has been criticised for not having separated waste collection charges for householders (either as a lump sum or volume- or tonnage-based) from the local council tax, and thereby missing an opportunity to provide more direct economic incentives for households to reduce waste (OECD 2002). In Sweden, the national average annual waste collection charge per person is about SEK 500 (c. EUR 54) (RVF 2003b).

Regarding the adoption of positive economic incentives in the form of *subsidies and grants*, there is a considerable difference. The UK Recycling Credit scheme (payable from local Waste Disposal Authorities to Waste Collection Authorities for reduced amounts of waste sent to landfill) introduced in 1992 was followed by the Landfill Tax Credit Scheme in 1996 (distributing tax revenue to various environmental projects), and in 2001 the WRAP set in place a number of funds for R&D funding, technology demonstration and dissemination, financial guarantees and project funding. The Waste Implementation Programme (WIP) further introduced competitive funding for local authorities for recycling projects, and has also encouraged a set of pilot schemes for local authorities to provide positive economic incentives for consumers to reduce waste and recycle more (DEFRA 2005g). However, all these funding instruments have been introduced in a context where both local government and the waste management industry has complained that the basic grant from the central government budget to local authority waste management has been consistently insufficient (see e.g. LGA 2004a). In Sweden about EUR 700 million was spent on the Local Investment Programme, and 7% of the resources were allocated to waste-related projects. The data suggests that the subsidies and grants provided by the UK government could be somewhere between five- to ten-fold those of the Swedish government.

England has also made use of *tradable permit systems* as an economic instrument, whereas Sweden has none in place. It has been claimed that its implementation of the packaging producer responsibility through the system of tradable Packaging Recovery Notes is unique (OECD 2002) and the annual value of trades is about GBP 26 million. Tradable permit systems have also been discussed as measures for implementing the producer responsibilities for electric and electronic equipment waste and for end-of-life vehicles respectively, but no concrete proposals were put forward. The other English system, the Landfill Allowance Trading Scheme, was introduced in April 2005 as a response to the Landfill Directive targets (see above). It facilitates trading, borrowing and banking of grand-fathered landfill allowances among local authorities. This instrument is analysed as a case study in chapter 7. In Sweden, the idea of tradable certificates for recycled plastics has been raised in the policy debate (Statens Offentliga Utredningar 2005a).

Finally, *deposit-refund systems* are a key feature in Swedish municipal waste policy while missing in England. The system for glass bottles appears to have emerged from breweries' initiative, while the ones for cars and aluminium cans were introduced through legislation as early as in 1975 and 1982 respectively. These, together with the one for plastic bottles in 1991, facilitated the introduction of producer responsibility and possibly made consumers more aware of recycling in general.

Considering the group of economic instruments as a whole, Sweden has opted for taxes and deposit-refund systems, while the UK has made greater use of tradable permit systems and subsidies and grants. It can be argued that in England there seems to be a tendency to, within the economic instruments group of EPIs, add instruments that help creating and strengthening markets (WRAP), enable actors to respond to market incentives and build capacity (WIP, Landfill Tax Credit Scheme), and fulfil regulatory obligations in more efficient ways (LATS, Packaging Recovery Notes). In Sweden, on the other hand, more market interventionist instruments have been introduced (higher tax rate and non-recycled revenue of landfill tax, deposit requirement on consumers). In general, England seems to prefer using economic incentives more towards the positive end of the spectrum, while Sweden seems to prefer incentives towards the negative end. The marked increase of the UK landfill tax over the next few years, however, suggests that the UK may be moving away from economic efficiency as a necessary justification for environmental taxes and charges, towards recognising also the effectiveness of such instruments in changing behaviour. While the landfill tax was originally based on estimates of negative externalities, the rationale of setting the tax rate was then modified to incentivise landfill diversion on a larger scale.

6.3.4 Liability and damage compensation

Within the liability and damage compensation EPI category, *producer responsibility*[52] was found to be the only relevant sub-category. Only mandatory and legally binding producer responsibility schemes were included here. As stated above, Sweden has been a forerunner with this kind of EPI, starting the process in 1994 and subsequently responding to the EU directives (see **Table 28**). The principles in the Swedish schemes (for packaging, newspaper, car tyres, end-of-life vehicles, and electric and electronic equipment) are that the producer should ensure, together with local authorities, that collection systems are in place, households are informed, and options for recycling, re-use or other environmentally acceptable disposal exist. The mechanism for translating these principles into practice are the quantitative targets for recycling or reuse. In England, meanwhile, only two schemes were implemented in 1995-2005; for end-of-life vehicles and for packaging (through the tradable permit system) (see **Table 29**). The producer responsibility for electric and electronic equipment was much delayed, and the regulations did not enter into force until 2007. It is thus clear that Sweden has been more active with this EPI.

[52] According to OECD (2001b), an extended producer responsibility scheme can be voluntary, negotiated, mandatory, or market-driven. Instruments that implement extended producer responsibility are identified as: product take-back requirements, deposit-refund schemes, advance disposal fees, material taxes, upstream combination of tax and subsidy, standards for minimum recycled content of products, leasing, and servicizing.

Table 28. Sweden: liability and damage compensation

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
LIABILITY AND DAMAGE COMPENSATION											
Extended producer responsibility	Producer responsibility: packaging – recycling targets for 2001 (1994-) *(SFS 1994:1235)*		Amended; recycling targets for 2005 *(SFS 1997:185)*								
	Producer responsibility: newspaper – collection target for 2000 (1994-) *(SFS 1994:1205)*										
	Producer responsibility: car tyres – non-landfill disposal target (1994-) *(SFS 1994:1236)*										
				Producer responsibility: cars – recycling and reuse targets for 2006 and 2015 *(SFS 1997:788)*							
							Producer responsibility: electric and electronic equipment – no target *(SFS 2000:208)*				

Table 29. England: liability and damage compensation

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
LIABILITY AND DAMAGE COMPENSATION											
Extended producer responsibility									Producer responsibility: End-of-Life Vehicles producer responsibility (*End-of-Life Vehicles Regulations 2003*)		

6.3.5 Education and information

As for education and information, the main responsibility for promoting household recycling lies at the level of local authorities. Nevertheless, both countries have introduced nation-wide public campaigns to promote recycling and support local information initiatives, both generally and in relation to specific waste streams (batteries and nappies) (see **Table 30** and **Table 31**). The UK appears to have a more diverse approach, though, by introducing a range of technology dissemination and business advice instruments under the WRAP and WIP programmes. In Sweden, such efforts are missing or administered by other actors than the national government. Ecolabelling and publicity of sanctions for non-compliance as information instruments were not considered here, due to their wider applicability than the municipal waste policy area.

Table 30. Sweden: education and information

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
EDUCATION AND INFORMATION											
Public campaigns							'Sopor.nu' – website with consumer information on recycling				
								'Hem till holken' – consumer information campaign on batteries collection			
Technology diffusion	(None)										

Table 31. England: education and information

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
EDUCATION AND INFORMATION											
Public campaigns	Rethink Rubbish										
						Recycle Now campaign (WRAP)					
						Home Composting Scheme (WRAP)					
						Real Nappy programme (WRAP)					
							Recycled Products Guide (WRAP)				
Technology diffusion						Materials Programme (WRAP): glass, Plastics, Paper, Wood, Organics, Aggregates					
						ROTATE (WRAP)					
									New Technologies: Supporter Programme (WIP)		

6.3.6 Voluntary approaches

The inventory results show that two negotiated agreements have been adopted in England (recycled content of newspapers, recycling of direct mail), as well as a unilateral commitment of the British automotive industry on recycling of end-of-life vehicles (see **Table 32**). In Sweden, none has been introduced in the municipal waste policy area as defined in this paper, but there are two negotiated agreements for commercially and industrially generated waste streams: office paper (1997) and construction and demolition waste (1997). Being a relatively new type of EPI to England (in comparison with for example the Netherlands) they represent an important contribution to the diversity of the mix. Furthermore, they are an important alternative, or precursor, to mandatory producer responsibility.

Table 32. England: voluntary approaches

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
VOLUNTARY APPROACHES											
Unilateral commitments			Automotive Consortium on Recycling and Disposal								
Negotiated agreements						VA with Newspaper Publishers Association					
									VA with Direct Marketing Association		

6.3.7 Management and planning

Finally, the kinds of *management and planning* instruments considered in this inventory are national requirements or encouragement of local waste plans and strategies[53]. Both have statutory requirements for local plans (see **Table 33** and **Table 34**), but the UK has

[53] Note that national waste strategies and plans fall outside the definition of instruments in this paper, since they define objectives and targets and/or represent a form of internal organisation or regulation. England has prepared several national waste strategies in 1995-2005 (see chapter 5, Table 18).

introduced new guidance and emphasised more strongly the need to ensure they are in line with landfill and recycling targets to a further extent in the time period studied.

Table 33. Sweden: management and planning

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
MANAGEMENT AND PLANNING											
Local plans	Statutory municipal waste regulations and waste plan (1991-) *(Env Code, 15 ch., 11 §)*										

Table 34. England: management and planning

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
MANAGEMENT AND PLANNING											
Local plans	Statutory Waste Recycling Plans										
	Statutory Waste Local Plan										
						Non-statutory municipal waste strategy					
											Statutory Joint Municipal Waste Management Strategy in two-tier areas (*Waste and Emissions Trading Act 2003*)

6.3.8 Summary of findings

To summarise, the results of the inventory of instruments show that both the size and the diversity of the municipal waste instrument mixes have increased in both England and Sweden, but particularly in England. The number of EPIs has risen dramatically over the last decade. All six EPI categories have been introduced by 2005 in England and five in Sweden (all except VAs). The number of economic instruments has increased in both countries, especially positive economic incentives in England in the form of grants and subsidies. Two tradable permit systems were also introduced in England in the studied period. Finally, a range of new information and education programmes were launched in England in the 2000s. Meanwhile, Sweden has not adopted as many new EPIs, and have also stuck to the command-and-control category to a larger extent.

In addition to the six categories defined by the OECD, *non-statutory targets* (without a VA) may be increasingly used as alternatives to other instruments. For example, Sweden has recently adopted recycling targets for the food industry and catering industry, indicating that policy instruments may follow if they are not met (see Regeringens proposition 2005a, p. 25). The interviews held in both countries suggested that targets are taken seriously by all actors. This means that the classification of targets as a form of public policy instrument needs to be reconsidered by policy analysts (see section 2.2). An increased use of *statutory* recycling targets for local authorities has also been observed in Europe and the OECD (OECD 2001b; CIWM 2005).

It should again be emphasised that the mere existence of an instrument in nominal terms does not inform us about its relative (perceived) significance, impact and effectiveness. However, the inventory presented in **Appendix V** (Tables V-A and V-B) suggests that England has been successful in responding to the quest for new EPIs and diversifying its instrument mix, whereas Sweden has been so to a lesser extent. In contrast to the trends for environmental policy as a whole described above, there is thus more of a rapid revolution in England compared to a steady evolution in Sweden in the waste field (c.f. Jordan, Wurzel et al. 2003a). It is also clear that EPI creativity and freedom to try innovative approaches exist at the national level rather than the EU level. The review of EU waste-related directives by Rittberger and Richardson (2003) showed that there were very few or no economic and suasive instruments specified in directives, but mostly command-and-control instruments in the waste policy field (as opposed to the water and air pollution directives).

In the introductory chapter (chapter 1) it was stated that diversification of the environmental policy instrument mix could be a sign of depoliticisation of instrument choice. It was then argued in chapter 2 that the *category* of the instrument may be perceived as less important and politically controversial than various *design properties* of an instrument (see Daugbjerg 1998a), of which *coerciveness* was seen as the most crucial one. Therefore, we need to examine not only the use of different categories but also the coerciveness of different instruments in the mix, i.e. the extent to which it is accepted that the state constrains private behaviour.

6.4 The coerciveness of instruments in the mix

What chronological sequencing patterns can be observed in terms of the relative coerciveness of waste policy instruments in Sweden and England? For this analysis the identified instruments mapped out in **Appendix V** were classified according to the typologies proposed by Doern and Wilson (1974 in Doern and Phidd 1983) and van der

Doelen 1998), in **Table 35** and **Table 36** (see Table 11 and Table 12 in chapter 4 for reference). As described in chapter 4, some adaptation of these typologies was needed[54].

With regards to the *overall mix* of municipal waste instruments identified here, a categorisation according to Doern and Wilson's typology reveals that Sweden has tended towards choosing more coercive instruments than England, i.e. more 'regulation' than 'exhortation' or 'expenditure' (measured in nominal terms) (see **Table 35**). Many 'regulation' instruments have been in place for a long time, while the instruments in the other two instrument categories were introduced more recently. In England, there is a greater spread among the less coercive and more coercive instrument types, and the data does not suggest that softer exhortation instruments have been, on the whole, introduced before expenditure and regulation instruments. At the overall level, the fit with Doern and Wilson's (1974) 'minimal-coercion' pattern is therefore poor. Plotting in the Swedish instruments indicates that coercive instruments were put in place first, followed by less coercive instruments to enhance the achievement of policy objectives, i.e. the opposite pattern to that predicted by the authors. In England, there seems to be no unambiguous direction of the sequencing at all. However, the effect of the relatively short timescale chosen here and the difficulties involved in defining when a policy issue is 'new' need to be considered when making these conclusions.

[54] In addition to the adaption described, the inventory exercise also raised some issues regarding the classification of tradable permit systems or deposit-refund systems. In Doern and Phidd's typology, deposit-refund systems have here been classified as a 'regulation' since they impose an opportunity cost on consumers. Tradable permit systems have also been put into this category since they in this case have introduced obligations (packaging recovery levels and landfill allowances) rather than rights. In van der Doelen's typology, deposit-refund systems have been classified as 'repressive economic control' and tradable permit systems have been classified as 'repressive judicial control' based on the same rationales. Note that if the tradable permit systems in this case had been auctioned rather than grandfathered they could have been classified also as 'repressive economic control'.

Table 35. A 'minimal coercion' pattern?

Exhortation	Expenditure	Regulation	Public ownership
SWEDEN			
***Information*: Public recycling campaign 2001; Batteries campaign 2002**	***Subsidies and grants*:** LIP 1998-2002; Investment grant for sorting facilities 2004	***Deposit-refund*:** cars 1975; aluminium cans 1982: plastic bottles 1991. ***Taxes and charges*:** Charge on imported aluminium cans 1983; Batteries charge 1987; Landfill tax 2000. ***Producer responsibility*:** packaging, newspaper, car tyres 1994; end-of-life vehicles 1998; electric and electronic equipment 2001. ***Permitting*:** waste management permitting pre-1995; tighter standards landfill 2001; tighter standards incineration 2002. ***Bans*:** landfill bans 2002 and 2005	**(None)**
ENGLAND			
***Information*: Public recycling campaigns 1994 and 2000; technology dissemination 2000. *VAs*: cars 1997; newspaper 2000; direct mail 2003**	***Subsidies and grants*:** Recycling credits 1992; Landfill Tax Credit Scheme 1996; WRAP schemes 2000; NWMRF 2003; WIP schemes 2004; WPE grant 2005; BREW programme 2005	***Taxes and charges*:** Landfill tax 1996. ***Tradable permit systems*:** PRNs 1997; LATS 2005. ***Statutory targets*:** SPSs 2001. ***Producer responsibility*:** end-of-life vehicles 2003. ***Permitting*:** waste management permitting pre-1995; tighter standards landfill 2002; tighter standards incineration 2002	**(None)**

Considering the overall mix in van der Doelen’s typology (see **Table 36**), his hypothesis could be interpreted to mean that one should expect a reasonable balance between the ‘give’ (stimulative) and ‘take’ (repressive) sides. It can again be found that Sweden has adopted comparatively more repressive instruments. There is arguably a greater balance in England’s mix on the whole. Regarding ‘economic control’, England has introduced more positive economic incentives than Sweden. Regarding ‘judicial control’, England has also added some stimulative instrument variants in addition to traditional repressive regulation, while Sweden has introduced on a wider basis producer responsibility as a newer form of repressive judicial control.

Table 36. A 'give-and-take' strategy?

	Stimulative	Repressive
SWEDEN		
Communicative control model	***Information*: ** Public recycling campaign 2001; Batteries campaign 2002.	N/A
Economic control model	***Subsidies and grants*: ** LIP 1998-2002; Investment grant for sorting facilities 2004	***Deposit-refund:*** cars 1975; aluminium cans 1982: plastic bottles 1991. ***Taxes and charges*: ** Charge on imported aluminium cans 1983; Batteries charge 1987; Landfill tax 2000.
Judicial control model	(None)	***Producer responsibility*: ** packaging, newspaper, car tyres 1994; end-of-life vehicles 1998; electric and electronic equipment 2001. ***Permitting*: ** waste management permitting pre-1995; tighter standards landfill 2001; tighter standards incineration 2002. ***Bans*: ** landfill bans 2002 and 2005
ENGLAND		
Communicative control model	***Information*: ** Public recycling campaigns 1994? and 2000; technology dissemination 2000.	N/A
Economic control model	***Subsidies and grants*: ** Recycling credits 1992; Landfill Tax Credit Scheme 1996; WRAP schemes 2000; NWMRF 2003; WIP schemes 2004; WPE grant 2005; BREW programme 2005	***Taxes and charges*: ** Landfill tax 1996.
Judicial control model	***VAs*: ** cars 1997, newspaper 2002, direct mail 2005	***Tradable permit systems*: ** PRNs 1997; LATS 2005. ***Statutory targets*: ** SPSs 2001. ***Producer responsibility*: ** end-of-life vehicles 2003. ***Permitting*: ** waste management permitting pre-1995; tighter standards landfill 2002; tighter standards incineration 2002

Examining the empirical evidence for these two hypothesised patterns at the *overall level* of the municipal waste policy mix as identified in this study thus suggests that Doern and Wilson’s hypothesis is not supported either in Sweden or England, while van der Doelen’s hypothesis may have some bearing for explaining the pattern in England. However, analyses at this overall level do not take into account the fact that there may be a rather eclectic mix of policy objectives and target groups preventing a clear and coherent government strategy, à la ‘minimal-coercion’ or ‘give-and-take’ (cf. Gunningham and Grabosky 1998). There is a need to focus on more specific sequences in *sub-mixes* or packages of EPIs. Analysing instruments for landfill diversion provides illustrative examples.

In Sweden, landfill site permitting was in place before 1995. In 2000, the landfill tax was introduced, followed by significant tax rate increases in 2002 and 2003. In 2001, the implementation of the EU Landfill Directive also meant more stringent emission and process standards, with the potential consequence of landfill closures. Finally, in 2002 and 2005 bans on landfilling of sorted combustible waste and organic waste were introduced (together with a temporary system for exemption licensing). Among these instruments there is thus a pattern of increasing coerciveness, through repressive judicial control and repressive economic control. There do not seem to have been a great deal of stimulative instruments for increased materials recycling and biological treatment accompanying these instruments, though. Less than a tenth of the waste-related Local Investment Programme projects were concerned with recycling and composting of household waste (Swedish Environmental Protection Agency 2004, p. 10). The waste incineration tax will improve the relative prices for materials recycling and biological treatment, but indirectly so and through a repressive mechanism. Thus, a rather consistently coercive mix of instruments appears to have been maintained, suggesting that neither Doern and Wilson’s nor van der Doelen’s patterns match with this case.

In England, on the other hand, a pattern resembling a ‘give-and-take’ strategy can be observed. Like in Sweden, landfill sites were subject to permitting procedures before 1995. From 1992 onwards, increased recycling of municipal waste has been stimulated through the Recycling Credits scheme. To increase the speed of diversion, the landfill tax was introduced in 1996. However, with its hypothecated revenues it is less repressive than the Swedish landfill tax. Indeed, hypothecated taxes in themselves represent ‘give-and-take’ parcels of policy. To respond to the challenging EU Landfill Directive targets, the WRAP and the WIP programmes were launched in 2000 and 2002 respectively. These contain a range of stimulative instruments, both of communicative and economic kind. In 2002, landfill permitting was then made more stringent as a result of the EU Landfill Directive. Another repressive instrument followed in 2005 with the launch of the Landfill Allowance Trading Scheme. However, as a tradable permit system, it is less coercive than statutory targets for example, since it allows greater flexibility for the target actors on how to achieve their allowances (reducing landfill or purchasing additional allowances).

To sum up, these results show that Sweden has a more coercive mix of instruments in use, while England has mixed less and more coercive instruments to a greater extent. In line with van der Doelen’s argument, there thus appears to be less of a need to legitimise policy in Sweden and target actors accept effectuating (i.e. repressive) instruments to a greater extent. This issue will be further discussed in the concluding chapter. Another interpretation could be that the purpose with stimulative instruments is not only to *legitimise* – and ensure continued political support – but also to *enable* actors to respond to repressive instruments in a more effective way. Whether this difference in the degree of coerciveness is in spite of, or because of, the fact that municipal waste generation and management problems are more urgent in England is another question. Other underlying reasons for these different patterns of coerciveness in Sweden and England – such as a statist political culture – will be discussed in the concluding chapter 9.

This analysis also raises a methodological issue; how to define and measure coerciveness. Both Doern and Wilson and van der Doelen implicitly define coerciveness in terms of *instrument types*. However, it was argued in chapter 2 that this is too crude and that there can be significant variation within a given instrument type (Daugbjerg and Svendsen 2001; Macdonald 2001). For example, taking the landfill taxes in Sweden and England as equally coercive would be mistaken. It is suggested here that further inquiries into coerciveness would benefit from comparing a smaller range of instruments and unpacking the notion of coerciveness. There are several relevant indicators of coerciveness, for example:

- *nature of the target group* – regulating private behaviour of individuals and citizens through various EPIs could be seen as more coercive than the behaviour of firms and organisations;
- *net transfer or resources* – if a target actor perceives a net benefit resulting from an EPI (such as financial resources, valuable information, or legal rights), it is likely to be seen as less coercive than if a net cost is perceived (see Linder and Peters 1989; Bressers and O'Toole 1998; Daugbjerg 1999);
- *level of ambition* – this relates to the extent of the intended (positive or negative) behaviour change and how demanding and intrusive it is perceived as (e.g. the stringency of an emission standard, the rate of a tax) (see Bressers and O'Toole 1998; Linder and Peters 1989);
- *universal application or individual adaptation* – universal application could be seen as more coercive since it allows less flexibility for the subjects of the EPI (see Daugbjerg 1999, p. 170); and
- *level of precision* – if an EPI specifies both a goal to be achieved and how that goal is to be achieved it is likely to be considered to be more coercive (see

Linder and Peters 1989; Bressers and O'Toole 1998; Daugbjerg 1999; Macdonald 2001).

The existence of credible enforcement and sanctions is also likely to affect the perceived coerciveness of an instrument.

6.5 Effectiveness of the instrument mixes

There is thus some evidence of patterns in the instrument mixes, but establishing whether they are the result of a coherent government strategy or a ‘kitchen-sink’ approach, based on satisficing various interests rather than optimising goal achievement (Gunningham and Gabrosky 1998), is another question. Such explanations of EPI choice will be further addressed in chapter 9. Meanwhile, the *effect* of the policy instruments on the waste problem can be examined. Does the diversity and/or coerciveness of the instrument mix matter for the effectiveness in solving the municipal waste problem?

The statistics on the national waste problems presented in chapter 5 (section 5.2) showed that while Sweden generated much less municipal waste in 1995 (380 kg/person) than the UK (496 kg/person), the relative increase by 2003 is of the same magnitude as in the UK (24% and 23% respectively). In relative terms and measured over this time period only, there is thus no evidence that a more coercive mix such as Sweden’s results in decreasing or stabilising *waste generation*. Importantly, most of the instruments have been focused on improving waste management rather than waste prevention, but it is nevertheless clear that any ‘spill-over’ effects on waste generation have been marginal at best.

Looking at *landfill* as the least preferable waste management option, though, there is a clear difference in that the UK has experienced an increase in landfilled amounts of 12% in the period 1995–2003 while Sweden has witnessed a reduction by 53% (see **Figure 5** and **Figure 6** in chapter 5). The Swedish landfill tax and landfill bans have thus been more effective than the English landfill tax and other instruments over this time period. However, drawing stronger conclusions about the relationship between coerciveness and effectiveness requires more analysis of drivers other than policy instruments (consumption trends, technology development, etc.), as well as disaggregated evaluations of individual EPIs.

Assessing effectiveness also involves comparison against stated goals. To what extent have the quantified policy targets set by the English and Swedish policy-makers over the time period (see **Table 19** and **Table 20** in chapter 5) been achieved? Looking first at municipal *waste generation*, England has not set a target whereas Sweden has set a target not to increase the total amount of waste. The Swedish target is thus not quantified or related to baseline year, but implies 0% growth. Clearly, this target has not been reached for the municipal waste stream.

Regarding targets set for *landfill*, the Swedish target stipulated a 50% decrease from 1994 to 2005. Although this applies to all waste except for mining waste, it has been reached with a large margin for the municipal waste stream, namely a reduction of 85% (from 1.38 million tonnes in 1994 to 0.21 million tonnes in 2005) (see **Figure 6** in chapter 5). For England, a target was set in 1995 to reduce the proportion of controlled waste (i.e. municipal, industrial and commercial waste) consigned to landfill from 70% to 60% by 2005. Measuring the achievement is more difficult since the term controlled waste is no longer used and 1995 data are not available, but available data suggest that the landfill of controlled waste was reduced from 59% in 1998/9 to 54% in 2002/3[55]. This suggests that the target, seen by many as too unambitious, has been reached. The achievement of the new generation of landfill targets in England, emerging from the EU Landfill Directive, is likely to be difficult though (National Audit Office 2006a). These targets are discussed further in the next chapter.

For *recycling and energy recovery*, a target was set for England in 1995 to recover (including material recycling and energy recovery) 40% of municipal waste by 2005. It was not met (DEFRA 2006). In 2003/04 all non-landfill waste management options together managed only 28% of municipal waste (DEFRA 2005d). Neither was the 1995 target to recycle or compost 25% of household waste by 2000 met. The recycling and

[55] Controlled waste includes municipal waste and industrial and household waste. Landfill of municipal waste decreased from 84% in 1996 to 72% in 2003 (see chapter 5). These figures can be added to industrial and commercial waste. According to the National Waste Production Survey available on the Environment Agency website, the amount of industrial and commercial waste landfilled in 1998/9 was 34.748 million tonnes and in 2002/03 30.062 million tonnes. The total industrial and commercial waste produced in these two years was 68.8 million tonnes and 67.9 million tonnes respectively.

composting level reached in 2000/01 was only 12%. This means also that it is likely that the future targets for recycling and recovery in 2010 and 2015 will be difficult to meet. Sweden did not introduce a target for recycling until 2005, when it was stated that by 2010 at least 50% should be recycled through material recycling and biological treatment. In 2005, the level was 44%, suggesting that the target was rather modestly set and could be reached. Overall, it appears that more ambitious targets (given the current state of the waste problem) have been set for England than Sweden, but that they have not been matched by effective instruments that have secured their achievement

In summary, it seems that Sweden's more coercive waste policy instrument mix has been more effective in steering waste management up the waste hierarchy, beside from increased prevention. However, other factors such as technology development and market trends need to be controlled for to establish causal relationships. England has not only failed to tackle the waste problem in an effective way, but also failed to reach many of its own policy targets for landfill reduction and increased recycling. Despite this, new targets were proposed in the latest 2006 waste strategy review (DEFRA 2006).

What new EPIs have been suggested for the future in Sweden and England? In Sweden, the government and Swedish EPA have argued that, due to the high level of policy activity in the late 1990s and early 2000s, there is now a need to focus on implementation and effectiveness, rather than to introduce new instruments (Naturvårdsverket 2005). Several of the interviewees argued that a 'saturation point' had been reached, at least for now. There is also wide agreement though that the focus now needs to be shifted to waste generation, but so far few concrete instruments have been suggested. This shift of focus is occurring also at European level (Tromans 2001, p. 133) and in England. The 2006 English waste strategy review promoted waste minimisation, although without quantified targets or coercive instruments (DEFRA 2006). Much of these latest strategic intentions are concerned with removing barriers to landfill diversion, by simplifying the regulatory and permitting system for waste management facilities and increasing the level of public funding for investments in new capacity. In other words, there is a focus on policy coordination and clearing up the 'policy mess' (cf. Sorrell and Sijm 2003). But also some new NEPIs are discussed; new producer responsibility schemes, new VAs, and information and awareness campaigns.

6.6 Conclusion

The review of the overall EPI mixes in the two countries suggests that both countries have diversified their use of instruments, and in that sense been successful to some extent with the NEPI quest. For environmental policy as a whole, Sweden particularly has diversified its mix. It was found that a large number of new taxes and charges and voluntary approaches were introduced in both countries in the 1990s, while in the 2000s there has not been such a rapid growth. There are no signs that the use of subsidies and grants have decreased. As emphasised by other students of instrument choice, however, the adoption of NEPIs has not meant that traditional regulation has become less common or important. Instead of a 'fixed sum' of government intervention shared between different EPIs, the overall level of intervention may have grown in the environmental policy field. When studying the spread of NEPIs, this is an equally important question that needs to be further explored. Overall, the results of the review presented here seem to be consistent with the OECD's assessment of the EPI mixes in their environmental performance reviews. The UK mix of EPIs was described in 2002 as having become "more balanced, with a greater role given to economic instruments in recent years, and continued effective use of regulation and land use planning" (OECD 2002, p. 151). Sweden was praised by the OECD in 2004 as a forerunner in the use of economic instruments, but that more attention was needed to the efficiency of instrument mixes (OECD 2004b, p. 31).

It can be concluded from this study that the increasing diversity of EPIs generally is reflected also in the instrument mixes adopted at the national level for municipal waste in Sweden and England between 1995 and 2005. In particular in England, the diversity dramatically increased, with numerous new instruments in the late 1990s and early 2000's emerging from revised waste strategies. Both countries have made use of economic instruments to a fairly large extent, with Sweden tending to use taxes, charges and deposit-refund systems and England tending to use subsidies, grants and tradable permit systems. This raises the issue whether instruments for market intervention are generally more accepted in Sweden, while instruments for market creation and reduction of transaction costs are generally preferred in England. Sweden introduced producer responsibility as a liability instrument earlier and has included a larger number of waste streams, while England has adopted more VAs. Considering also new instruments adopted in the command-and-control and education and information categories, it could be argued that there has been a significant degree of instrument innovation in the municipal waste policy areas in Sweden and England. Whether this development of increased diversity is a result of general government commitments to use of alternative EPIs, or other factors (e.g. party politics, policy network relationships, transfer of policy ideas) remains an issue for discussion in chapter 9, however.

It was further found that Sweden has a generally more coercive instrument mix for municipal waste policy than England, using the typologies of Doern and Wilson (1974) and van der Doelen (1998). The 'minimal-coercion' hypothesis of Doern and Wilson regarding the sequencing of instruments in a mix did not match with the patterns in either country. Instead, Sweden seems to have opted for relatively coercive instruments, from an early stage. There seems to be more of a 'give-and-take' pattern, as proposed by van der Doelen, in the instrument mix in England, though. These theories on coerciveness would benefit from incorporating the (co-)evolution of the policy problem character, which neither of the two discussed here did satisfactorily. For example, does a gradual 'solving' of a problem increase the public support for even more coercive instruments to be introduced, or are 'softer' instruments preferred? Also, the notion of coerciveness could be further unpacked, and a set of potential indicators were suggested for this purpose. In any case, it was found that the effectiveness of the policy instrument mix over the 1995-2005 time period may have been higher in Sweden. Although a thorough evaluation is required to examine the effects from policy instruments and other drivers of change separately, this raises the question whether coercive mixes are generally more effective than less coercive ones.

To conclude, this chapter has demonstrated that a trend of diversification is apparent both in the municipal waste policy field and in environmental policy more broadly. Thus, the quest for NEPIs has been successful, but whether it has been successful enough is another question. Whereas England has been more successful in increasing the diversity of EPIs to tackle the municipal waste problem, Sweden seems to have been more effective with its more coercive approach. This raises the question whether having a diverse mix of EPIs is really an end in itself or whether effectiveness should be the only goal. This complex question will be discussed in chapter 9.

Chapter 7

The England Landfill Allowance Trading Scheme

7.1 Introduction

The previous chapter described how the municipal waste policy instrument mix in England has grown highly diverse over the last decade. The question is whether this trend has been accompanied by EPI choice *processes* that encourage the adoption of new forms of EPIs, facilitated by systematic assessment of alternative instruments that strengthen procedural rationality in the policy process. This chapter will address this question through a case study of the process of developing the *Landfill Allowance Trading Scheme* (LATS) instrument. It is one of the most recent instruments introduced in England, entering into force on 1 April 2005. The purpose of this instrument is to tackle the country's landfill problem, by allocating allowances to local authorities for the amount of biodegradable municipal waste (BMW) that they can send to landfill each year. The total amount of allowances is reduced each year. The LATS also promotes more cost-effective reduction of landfill by allowing local authorities to trade allowances with each other, or bank or borrow between years. The LATS is thus a *tradable permit system* (TPS), hence an economic instrument and a so-called NEPI. At its launch, Environment Minister Elliot Morley stated that

> "[t]he LATS is an innovative and flexible approach which moves Government away from the old tools of command and control by offering an alternative to the regulatory system of inflexible targets" (DEFRA 2005e).

The aim of this chapter is to analyse the process leading up to the introduction of the LATS. To what extent was it characterised by procedural rationality? What political and institutional factors influenced the choice of a TPS? It will be argued that important factors influencing the choice of a TPS, as well as the relatively high degree of procedural rationality, included the external imposition of quantified policy targets, the high-cost nature of the 'landfill diversion' problem, the possibility to compensate the losers (in this case, local authorities), and also active promotion of NEPIs. However, it will also be argued that the debate and controversy in this EPI choice process was concentrated in the initial problem definition phase and in the phase of working out the detailed mechanisms of the instrument, as opposed to the selection of EPI *type*.

The next section (7.2) will provide some background to the problem and policy process, followed by a chronology of the major decisions in section 7.3. Based on this overview, the analytical framework of elements of procedural rationality developed in chapter 4 is then applied in section 7.4. After a summary assessment, the macro-, meso- and micro-level analytical framework is used to support the analysis of political and institutional factors influencing the process and the EPI choice outcome (section 7.5). Conclusions from this analysis are made in section 7.6, together with a brief review of the prospects of the LATS.

7.2 Background

As described in chapter 5 (section 5.2), there was an increasing awareness of the problem of high reliance on landfill in the early 1990s. In the mid-1990s, more than 80% of all municipal waste in England was sent to landfill (see **Figure 5** in section 5.2). An aspirational landfill reduction target had been set in 1995 to reduce landfill of controlled waste from 70-60% by 2005. The landfill tax was introduced in 1996 as a means to achieve this target. However, it was the 1999 *EC Landfill Directive*[56] Article 5(2) targets that provided a final impetus – as well as legal reasons – for the need to take more drastic action. These targets applying to Member States specified levels of diversion of BMW away from landfill towards waste treatment methods higher up in the waste hierarchy (i.e. incineration with energy recovery, materials recycling, biological treatment, or prevention). For the UK[57], these targets[58] are to reduce the amount of BMW landfilled to (a maximum of):

[56] Directive 1999/31/EC on the landfill of waste (agreed by the Council on 26 April 1999, entered into force on 16 July 1999). Note that the Landfill Directive also lays down procedures, standards and bans in relation to the permitting and management of landfill sites (see Articles 5.3, 6-14, Annexes I-III). These provisions are of a more regulatory than strategic kind and do not involve the instrument choice flexibility on the part of Member States that the Article 5.2 targets do.
[57] Countries like the UK that landfilled more than 80% of their BMW were given the possibility to seek derogation of the target years by four years, and the UK was granted this derogation (see further discussion below). The normal target years for member states are 2006, 2009, and 2016.
[58] Hereinafter these targets will be referred to as the landfill targets, although they apply to BMW only.

- 75% of the 1995 level by 2010
- 50% of the 1995 level by 2013
- 35% of the 1995 level by 2020.

Considering that the biodegradable fraction of municipal waste is around two-thirds (DETR 1999a, p. 12), the targets thus applied to a large proportion of municipal waste and would require significant changes to the English waste management system. Accordingly, the targets for landfill diversion were widely recognised as implying a 'sea change' in UK waste management, more so than in other Member States that were less reliant on landfill (see **Figure 7** in section 5.2 for a European comparison). For example, the waste management industry regards the Landfill Directive as

> "the single most important driver for our sector's future" (ESA, in HoC Select Committee on Environment Food and Rural Affairs 2005b, p. 80).

The targets imply, directly, a need to limit landfill of BMW and, indirectly, a need to divert this waste to other management options like incineration and recycling – all in the context of increasing household waste generation. Given the widespread resistance to large-scale incineration – which was seen as the only realistic diversion option in the short and medium-term – the targets raised tricky policy problems. It was clear that a mix of policy instruments was needed, not only to limit landfill but also to stimulate and ensure sufficient capacity of other waste treatment options. This case study, however, will look primarily at the instrument choice relating to the first part of the task, i.e. to limit landfill. Furthermore, this case study is delimited to the targets applying to England only and their implementation through the LATS instrument. The UK-wide BMW targets were subdivided among the devolved administrations and they have developed similar instruments[59].

In line with standard EU legislation, the Directive set these targets but "leaves the choice of the instruments to the Member States" (Commission of the European Communities 2005b, p. 5). Below, it will be shown that a range of alternative EPIs were

[59] The case study presented in this chapter is only concerned with the England LATS instrument. Scotland, Wales and Northern Ireland have all developed similar allowances schemes in response to the UK BMW targets; see Landfill Allowance Scheme (Scotland) Regulations 2005 (Scottish Statutory Instrument 2005 No. 157), Landfill Allowance Scheme (Wales) Regulations 2004 (Welsh Statutory Instrument 2004 No. 1490), and Landfill Allowance Scheme (Northern Ireland) Regulations 2004 (Statutory Rule of Northern Ireland 2004 No. 416). Important variations refer to the possibility to trade permits or not and how biodegradable content is calculated. There are plans to merge the English and Scottish systems from 2008 onwards and thereby increase the market for permits. For a comparison of the four schemes, see DEFRA (unpublished).

considered in the process, including command-and-control regulations for landfill gas capture, an increased landfill tax rate, bans on landfill, and statutory targets limiting landfill. The final selection of a TPS was supported by most stakeholders.

The *Department of Environment, Transport and the Regions* (DETR) (from 2001 onwards, the *Department of Environment, Food, Rural Affairs*, DEFRA) was responsible for proposing an instrument. Central government initially opposed the landfill targets as they were expected to lead to a ‘dash to incineration’, but were then concerned with implementing them as cost-effectively as possible. This view was shared by the *Environment Agency* (EA), who would be monitoring the instrument. Local government, represented mainly by the *Local Government Association* (LGA), was a key stakeholder, since they were responsible for waste management contracts and would make the day-to-day decisions on landfill diversion, as well as bear the immediate additional cost. The waste management industry, mainly represented by the *Environmental Services Association* (ESA), were also strongly critical of the landfill targets at first, but then had to accept that a major restructuring of the sector would take place. Finally, the professional association, the *Chartered Institute of Waste Management* (CIWM), united with the others in the opposition towards the targets, as did the manufacturing industry through the *Confederation of British Industry* (CBI). The only stakeholders welcoming the landfill targets were the incineration industry and environmental NGOs, such as *Friends of the Earth* (FoE) and the *Green Alliance*. The incineration industry, who would gain market share from the targets, was represented by the *Energy from Waste Association* (EfWA). This organisation merged with the ESA in 2001, as a reflection of the sector restructuring. The environmental NGOs, on the other hand, welcomed them on the premise that policies ensuring waste was diverted to recycling rather than incineration were introduced. Overall, there was a significant degree of agreement between many of the key interest groups in this process.

7.3 Key decisions and milestones in the process

The most important decisions and documents in the process are listed in the timeline in **Table 37** below. There were three distinct phases in this process; the decision on *policy targets* in 1990-1997, the selection of a TPS as an *EPI type* in 1998-2000, and the decisions made on detailed *design features* of the LATS in 2001-2005.

Table 37. Timeline of the LATS process

Year	EU Landfill Directive	Parliamentary and government decisions	Preparatory work, government consultation papers and consultant reports
1995			
1996			
1997			First Compliance Cost Assessment of the BMW targets
1998			Spring: DETR hosts workshop on implementation of landfill targets
1999	16 July: *Landfill Directive* enters into force		Consultant reports on possible instruments (ECOTEC) and economic analyses of landfill demand and a system of tradable landfill permits October: First consultation paper, *Limiting Landfill* Two seminars held
2000		*Waste Strategy 2000* announces a TPS has been selected	
2001	16 July: Transposition deadline		March: Second consultation paper, *Tradable Landfill Permits*
2002		Nov: *Waste & Emissions Trading Bill* introduced in the House of Lords	
2003		Nov: *Waste & Emissions Trading Act* given Royal Assent	August: Third consultation paper, *Landfill Allowance Trading Scheme* Sept-Oct: Six consultation seminars
2004		22 July: Regulation on Scheme Year and Maximum Landfill Amount for UK August: Provision allocation of allowances to WDAs December: Regulation on the England LATS	April: Publication of third consultation outcome and decisions made July: LATS roadshows held
2005		February: Final allocation of allowances to WDAs **1 April: LATS enters into force**	
2006			

The Landfill Directive was adopted by the Council in April 1999. However, the content of the final draft was settled before then. Work on what was to become the Landfill Directive started already in 1990 (Anon. 1995a), so the *first phase* of the LATS choice process thus started in the early 1990s when the UK government started negotiating on how to limit landfill. The proposed Directive went through several rounds in the EU institutions until agreement could be reached on the main provisions. The Commission, the European Parliament and some Member States pushed for tougher regulations on the management of landfill and some form of reduction targets. The UK was among the Member States most strongly resisting reduction targets (Anon. 1997e). The House of Lords Select Committee on European Communities (Sub-Committee D, Environment and Agriculture) inquiry into *Sustainable Landfill* in 1998 reflected the UK concern with landfill targets and expected large-scale diversion to incineration (see HoL Select Committee on European Communities 1998b; Anon. 1997a).

With the imminent finalisation of the formulation of the landfill targets at the EU level, a *second phase* started in the UK in 1998. The process of finding an appropriate instrument to achieve the targets, as well as transposing the targets into national legislation, was then initiated. DETR had begun thinking about instruments in late 1997 (see HoL Select Committee on European Communities 1998a, p. 72). In June 1998, the Labour government issued a consultation paper, *Less Waste, More Money*, for the development of a new comprehensive waste strategy for England and Wales (DETR 1998). Recognising the targets in the draft Directive, a first public statement about possible instruments was made;

> "the government is considering a number of options, such as limits on the amount of biodegradable waste that individual landfill sites can accept, or issuing permits to local authorities for the waste they can send to landfill, with a possible option of making those permits tradable" (para. 3.3.6).

Preceding this consultation paper was a workshop for key stakeholders hosted by the government in the spring of 1998 on how the landfill targets could be transposed and possible instruments designed. The government had showed interest in the idea of a TPS at this event, but targeted at landfill operators rather than local authorities (Anon. 1998).

In 1998 and 1999, DETR assigned a group of consultants to study various aspects of the Landfill Directive and the market for landfill in the UK. A key report coming out of this work was the report prepared in 1999 by ECOTEC consultants on *Policy Instruments to Implement the Proposed Landfill Directive BMSW Targets* (ECOTEC 1999). This report identified and examined a very wide range of possible instruments and concluded that phased introduction of a landfill ban, a higher landfill tax on BMW, and tradable landfill permits issued to landfill operators or local authorities were the most promising options. Also in 1999, DETR commissioned Enviros consultants to analyse landfill capacity demand and prices and to economically model a TPS for landfill[60].

In September 1999, the first of the three consultation papers on the choice of instrument for meeting the landfill targets was issued, *Limiting Landfill*, and two consultation seminars were held (DETR 1999a). Referring to the ECOTEC report, a short-list of five options was presented. The government's initial view was that a TPS was the better instrument option, but views were invited on whether it should target landfill operators (option 3) or WDAs (option 4) since the pros and cons of the two were "finely balanced" (p. 19). In the *Waste Strategy 2000*, it was stated that a TPS for WDAs had been the favoured option by a large majority of the consultees (over 70%). The strategy announced that such an instrument would be developed (DETR 2000a). The choice of *EPI type* had thus been formally made after about two years of deliberation.

With a decision made on the type of EPI to be used, a *third and final phase* started in 2001 when various design features of the TPS were addressed in the second consultation paper, *Tradable landfill permits consultation paper*. This paper invited views on the definition of the permits, allocation methods, trading rules, monitoring, and sanctions (DETR 2001). After consultees had submitted their views, the UK-wide *Waste and Emissions Trading Bill*[61] was prepared and presented to Parliament on 14 November 2002. In brief, it set out the powers and duties of allocating authorities (i.e. the Secretary of State) and disposal authorities in the devolved administrations, and set the scene for more specific regulations to come.

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[60] According to Enviros consultants, three reports were prepared for DETR: *Landfill capacity demand and prices* (1999), *Landfill permits* (with Aspinwall consultants, 1999), and *Explanation of modelling to estimate the economic benefits of tradeable landfill permits to meet the requirements of the landfill directive* (1999). Upon repeated requests, these reports were not released by Enviros, despite written consent from DEFRA, and have thus not been possible to include in the case study data.
[61] Waste and Emissions Trading Bill, House of Lords, Bill 3, 53-2.

While this bill was being debated in Parliament, the third and final consultation paper, *Landfill allowance trading scheme consultation*, was issued in August 2003 (DEFRA 2003b), together with a series of six consultation road shows. The draft LATS regulations were attached to this paper, which invited views on the 'detailed operation' of the instrument; the proposed modification of the allocation method, how the monitoring system should work, and levels of penalties for exceeding allowances. It also stated that the government intended to launch the LATS as soon as possible after the Bill had been adopted.

In November 2003, the *Waste and Emissions Trading Act*[62] was given Royal Assent, which meant that Article 5.2 of the Landfill Directive was not legally transposed until two years after the EU deadline. In April 2004, the outcome of the third consultation was published (DEFRA 2004a). At this point, the final design of the LATS was thus more or less clear to all stakeholders (except for the exact allowance allocation). In July, a second series of road shows were held to inform local authorities about the LATS. The *Regulations on scheme year and maximum landfill amount*[63] were also issued in July 2004, setting the landfill targets (and total level of allowances) for each of the devolved administrations in the UK.

Not until August 2004 were local authorities in England informed about the much awaited provisional *allocation of allowances* by the Environment Minister Elliot Morley. Before then, local authorities had to guess their allocations (based on the allocation formula established after the third consultation). The authorities then had the opportunity to comment, resulting in the final allocations being announced in February 2005 (DEFRA 2005c).

Finally, not until 3 December 2004 were the *Regulations on landfill allowances and trading scheme in England*[64] adopted, i.e. approximately a year after the draft regulations were presented. These regulations outline all the rules for trading, banking, borrowing, monitoring, reporting, and paying penalties. The regulations also formally

[62] Waste and Emissions Trading Act 2003, 2003 Chapter 33.
[63] Statutory Instrument 2004 No. 1936. The Landfill (Scheme Year and Maximum Landfill Amount) Regulations 2004.
[64] Statutory Instrument 2004 No. 3212. The Landfill Allowances and Trading Scheme (England) Regulations 2004.

announced that the LATS would enter into force on 1 April 2005. The launch of the scheme, which was one year later than originally planned, was preceded by another round of workshops in the winter of 2005. A minor amendment[65] of the Regulations was then made already in May 2005, when it was decided that the penalty levels for exceeding the allowance should be lowered to £150 from £200 per tonne BMW.

By 2005 the LATS instrument was thus complete in its design and implemented in practice. Its detailed design and mechanisms are described in **Box 1**. The basic idea of a tradeable permit system (TPS) is that principles of command-and-control regulation and economic incentive-based instruments are combined. A total level of emission or resource extraction permits is set and subdivided among the target actors, thus ensuring a certain environmental outcome. These permits can then be traded on a market, hence ensuring a cost-effective reduction by allowing actors with low marginal costs of reducing their environmentally damaging behaviour to sell surplus permits and *vice versa* (see Pearce and Turner 1990, pp. 110-119 for the economic theory of TPSs). Since it was expected that costs of reducing landfill diverged across local authorities in England, it was assumed that trading would decrease the total cost of meeting the landfill targets.

The '*permits*' in the case of the LATS refer to the right (or 'allowance'), expressed in tonnes BMW, for Waste Disposal Authorities (WDAs) to send BMW to landfill. WDAs are the branch of local authorities responsible for waste disposal and there are 121 of them in England[66]. The allowances were allocated by a 'grandfathering' approach that took into account historical levels of municipal waste generation, but also current levels of biodegradable waste landfilled. Annual allowances were set for the years 2005-2020 and gradually decrease in order to meet the aggregate England targets in the target years (2010, 2013, 2020). The allowances are '*tradeable*' in the sense that a WDA can bank and borrow allowances from itself between years, with some restrictions. It can also trade allowances with other WDAs on a monetary or non-monetary basis. DEFRA has set up an online bulletin to facilitate such a market operating efficiently. To make sure

[65] Statutory Instrument 2005 No. 880. The Landfill Allowances and Trading Scheme (England) (Amendment) Regulations 2005.

[66] Another branch of local authorities, Waste Collection Authorities, are responsible for the collection of waste. Some local authorities do not have separated collection and disposal functions. For an overview of the institutional arrangements in local authority waste management, see Davoudi (2000).

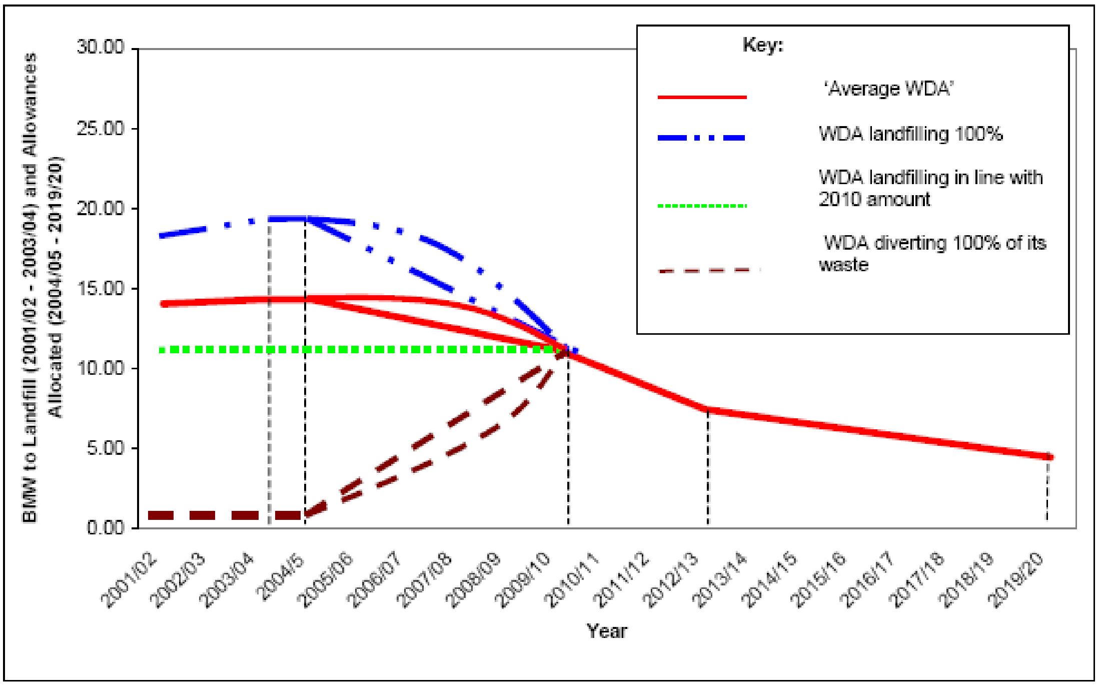
the LATS is effective, WDAs must report their landfill of municipal waste and estimated biodegradable content each year. If they exceed their allowances – after banking, borrowing or trading – the penalty is £150 per tonne BMW.

Box 1. Detailed description of the LATS

The scope of waste subject to the LATS is BMW, where **municipal waste** is defined as “waste from households, as well as other waste which because of its nature of composition, is similar to waste from households” (Article 2(b) of the Directive). The **biodegradable content** of MW (with biodegradable defined as “any waste that is capable of undergoing anaerobic or aerobic decomposition, such as food and garden waste, and paper and paperboard” in the Directive) is calculated through a general estimate of 68% as well as a specifically developed analytical tool, M-BEAM (see http://lasupport.defra.gov.uk).

In the case of LATS, the **total number of permits** (or **allowances** as they were renamed in 2003) allocated and tradable correspond to the England BMW tonnage targets in the target years (one tonne = one allowance). The England targets were set based on an apportionment of the UK aggregate BMW targets, which represent the percentage reductions stipulated by Article 5.2 of the Landfill Directive in relation to the 1995 base year. The maximum amount of BMW that can be landfilled in England is 11.2 million tonnes in 2010, 9.13 million tonnes in 2013, and 5.22 million tonnes in 2020. The principle for setting the total number of permits was thus completely target-led, as opposed to a calculation of the optimal level of landfill based on marginal social cost and benefit. Note that there is no ‘safety margin’ in the target years, i.e. the number of total allowances is equal to, not lower than, the target tonnages.

Holders of the allowances are the 121 Waste Disposal Authorities (WDAs) in England. The **allocation of allowances** was based on a **grandfathering** approach. The allocation formula took into account both historic and current levels of landfilling, as a result of a compromise between supporting those authorities with high landfill levels and rewarding those authorities which had already made investments in diversion. The figure below illustrates the effect of this compromise on authorities entering the scheme with different levels of landfill. The allocations for the target years (2010, 2013, 2020) are based on applying each WDA’s percentage contribution to 2001/02 (note that the EU Directive 1995 base year was not used due to lack of reliable data) England *overall municipal waste (MW) arisings* to the maximum amount BMW that can be sent to landfills from England in target years. This has the effect of rewarding those WDAs who receive a relatively low fraction of *biodegradable* MW in their waste, as a consequence of composting and recycling initiatives. It also rewards those authorities which have taken measures to divert BMW from landfill, compared with if allocations had been made on the basis of landfilled BMW rather than landfilled MW. The latter would have penalised those who had already reduced their reliance on landfill, by giving them fewer permits. However, to make the initial years’ (2005-2009) allowances a smoother transition for WDAs starting out with a high reliance on landfill, a base year was defined as the *total level of BMW landfilled* in 2001/02. In the figure below, this means that those WDAs end up on a trajectory somewhere between the red and blue lines. WDAs that had already diverted from landfill, meanwhile, end up between the red and brown line and will see an increase in allowances over the years until 2010, thus receiving a windfall profit. A ‘back-end loaded trajectory’ was then set for the first four years, which meant that allowances would not be reduced by an equal proportion each year but would be more generous in the first years. In the figure below, this ‘back-end’ loading is illustrated by the rounded curves instead of the straight lines in the period 2005-2010. For the interim years after the first target year 2010, allocation of allowances was based on equal instalments in absolute terms, implying a straight trajectory.



Note: This diagram assumes for simplicity reasons that all WDAs have the same level of waste arisings, with the purpose to highlight the effect of different levels of landfilling.
Source: DEFRA (2003b, p. 22, figure 1).

The *actual* tonnages landfilled, i.e. the performance in relation to the annual allowance, must be quarterly **reported** to the EA each year. At the end of each scheme year (running from 1 April to 31 March) there is a 6 months reconciliation period. If WDAs exceed their allowances a **penalty** of £150 per tonne is charged. The WDAs are also fined £1,000 if they fail to report actual tonnages landfilled.

The allowances are tradeable and flexible in three different ways, subject to certain rules. First, a WDA can **bank** its surplus allowances from one year to the next, except for the target years (2010, 2013, 2020) and the years preceding them (2009, 2012, 2019). Second, a WDA can **borrow** up to 5% of its next year's allowances, except for in the target years and the years preceding them. Third, they can be **traded** with another WDA, either in monetary terms, in non-monetary terms or on a futures basis. DEFRA has set up a bulletin to help WDAs announce trades and an electronic register to record allowances allocated and used (http://defraweb/environment/waste/localauth/lats/register). The register is publicly accessible, while the bulletin board and trade log are for WDAs and DEFRA use only. The trading log lists the price and quantity of every transaction to allow WDAs to monitor the market price, but does not reveal the trading parties.

Due to the six months reconciliation period, the **outcomes for 2005** in terms of trading patterns and actual tonnages landfilled are still unknown. However, a review of the publicly accessible electronic register in January 2007 revealed that 25 transfers of allowances between authorities had been registered and one borrowing. 17% of the 121 WDAs were involved in the transfers, which comprised in total 430,000 allowances and represented 3% of the total number of allowances. The price level was around £20 per allowance (A. Doran, *pers. comm.*). This trading outcome was higher than originally expected by DEFRA.

Sources: DEFRA (2003b), DEFRA (2005c), The Landfill (Scheme Year and Maximum Landfill Amount) Regulations 2004, Statutory Instrument 2004 No. 1936.

7.4 Procedural rationality in the LATS process

The LATS process as an EPI choice process thus spanned more than a decade, from the early 1990s to 2005. Below, an analysis is made of the extent to which the six elements of procedural rationality defined in the analytical framework in chapter 4 (see **Table 15**) characterised the LATS process; linear sequence of decision-making stages, clear policy objective and choice criteria, identification of several alternative instruments, systematic Regulatory Impact Assessment, consultation, and use of evidence.

7.4.1 A linear sequence of decision-making stages

As suggested by the chronology in **Table 37**, the LATS instrument choice process followed a linear and logical decision-making order according to the stages model. Within the EU Directive negotiations, the *problem definition* had been more or less established by the mid-1990s. The initial rationale for landfill targets was based on negative environmental effects in the form of leachates, visual disamenity, and the resource efficiency value in moving up the waste hierarchy. Over time, the emissions of landfill gas (primarily methane and carbon dioxide) came to be seen as the key problem. As an illustration, landfills released about 25% of the UK's methane emissions in 2001, which represents 2% of the world's total greenhouse gas emissions (DEFRA 2005f)[67]. By defining the landfill problem also a climate change problem, it became difficult for the UK government to dispute it (Anon. 1996). Still, there was considerable disagreement whether the best solution was to increase landfill gas capture or to reduce BMW input, which will be further analysed below. As mentioned above, the landfill problem was known in the UK before the EU Landfill Directive, but several interviewees were sceptical of the probability that a similarly forceful EPI would have been adopted based solely on national recognition of the problem.

> "Without the [Landfill] Directive, the business-as-usual approach would have gone on for much longer and the government wouldn't have interfered."

The *policy objectives* were then discussed and finally defined by 1999, in the form of the quantified BMW reduction targets. The Commission's original proposal in 1996 was to ban waste with a total organic content of more than 10% from landfill (Anon. 1996). Even when this TOC limit was relaxed to 20%, the two British EU Commissioners blocked the proposal. The new proposal in early 1997 was then to have a three-stage reduction of BMW in the form of timed targets (Anon. 1997b). After some negotiation on reduction percentages and target years, this approach was finally accepted by all Member States and decided upon in April 1999. The target that would apply to England could then be deduced by the policy-makers.

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[67] One tonne of BMW produces between 200 and 400 m^3 of landfill gas.

Discussion over possible instruments to implement the targets in England started in 1998, but *choice criteria* were also explicitly discussed from around this time. In the first consultant report on possible instruments, each instrument was systematically evaluated against eight criteria (ECOTEC 1999). The first consultation paper then listed six criteria taken from the Better Regulation Taskforce, some of which it used in the description of each of the five instrument options proposed (DETR 1999a).

A study of *possible instruments* was published in 1999. A TPS quickly emerged as the most interesting option, being considered more in-depth even before the study was published. However, the first consultation paper following the study still *evaluated* a shortlist of five possible instruments in relation to the criteria and invited views on which was most preferable. A *final decision* was then made after this consultation, when a TPS solution targeting WDAs was announced in the 2000 Waste Strategy (DETR 2000a.). After the instrument type had been decided in 2000, the process proceeded over the next five years by making more *detailed design choices* regarding allocation methods, trading rules, reporting, sanctions, etc.

As stipulated by the procedural rationality ideal, the process thus took place in a near perfect linear sequence of stages without significant iteration, with strategic decisions being made before operational decisions, and with a clear beginning and end. No interviewees had any major complaints about the set-up or logic of the process, but saw it as ‘straightforward’. The problem definition and policy objectives were finalised early on, by 1998. Since they were determined externally by the EU, the need for national consensus at each stage in process was precluded. They were non-negotiable once settled, so there was no point in re-questioning the rationale of the instrument in the ensuing debates. Furthermore, there was no visible attachment to a certain instrument before the problem had been defined, the policy objective set, and the criteria specified. Although there was early interest in a TPS instrument within DETR, the first consultation paper invited views on several instrument types and made no recommendations. Finally, the need to transpose the Directive meant that there was a deadline to the instrument choice process (although it was not met in the end).

7.4.2 Clear policy objectives and criteria

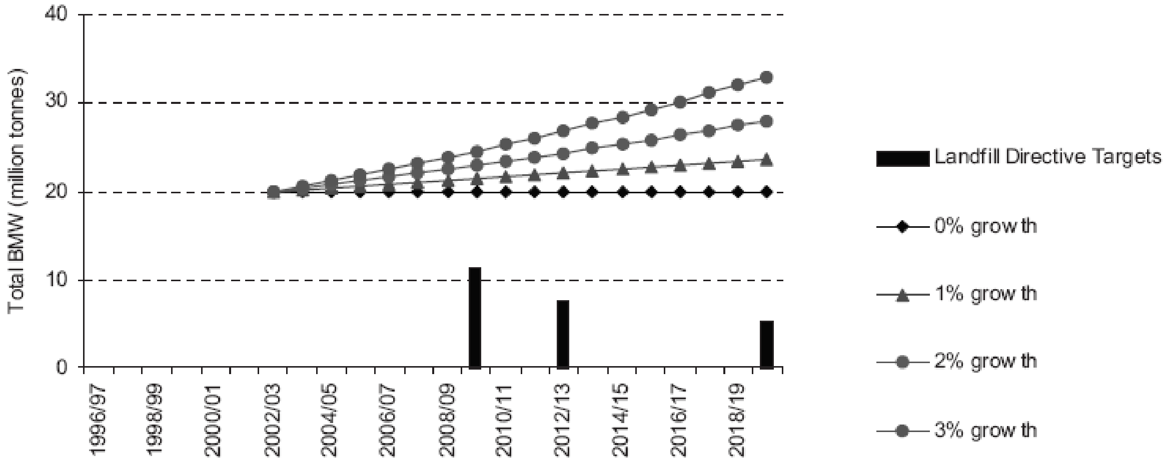
The previous section concluded that there were distinct and logically sequenced stages of setting policy objectives and choice criteria. The question is whether they were sufficiently clear, unambiguous and ranked, in order to serve as a good and transparent basis for instrument choice? As *policy objectives*, the landfill targets for England (shown in **Figure 10**, with projections of future BMW generation) were *quantified*, *precise* (in terms of targeting a defined waste stream), and had *fixed deadlines*. This meant that there was little room for ambiguity and interpretation by stakeholders as to the actual level of ambition. It also meant that there were clear benchmarks against which to model potential success of alternative EPIs. As described by one interviewee,

> "[the quantified nature of the targets] meant that it was very clear to everyone where we had to go and what policy had to achieve, and you could not really argue with that".

This could be compared to if the objective had been stated only in qualitative and open-ended terms, such as 'decrease landfill', when a formal or informal interpretation would be necessary.

Neither were there any major *direct* goal conflicts, since the targets did not specify where waste should be diverted *to*, i.e. incineration or recycling (or reduced waste generation). The debate on landfill diversion was vivid, but it did not have to take place within the LATS process. Much debate was thus defused due to the external specification of targets, that were in addition precise and quantified. However, clear and quantitative targets may not stimulate 'beyond compliance' either, exemplified by the decision of the UK government to avoid 'gold-plating' and not include a 'safety margin' when subdividing the national target into country targets, and later into local WDA allowances (a 1% safety margin was estimated to cost £23 million per year) (DEFRA 2003c, p. 7).

Figure 10. The England landfill targets



Source: HoC Select Committee on Environment Food and Rural Affairs (2005a, p. 21, table 3).

Regarding the instrument *choice criteria*, there were two reports that explicitly defined criteria and used them to systematically assess instrument options; the ECOTEC consultant report[68] that proposed a 'long-list' of instrument options in 1999 (ECOTEC 1999) and the subsequent first DETR consultation paper where these instrument options had been narrowed down to five (DETR 1999a). In the latter, the criteria included:

> "certainty in meeting the Directive's targets (including the targeting of the instrument towards its objectives); ease of administration and enforcement; transparency of the instrument; consistency in its application; proportionality in its effect (including environmental and economic effects); accountability" (p. 15).

It quickly became clear that *certainty* in meeting the targets – implying a minimum level of predictability of instrument outcome – would be the first-order criterion. The main reason was that it was expected that the European Court of Justice would fine the UK for breach of the targets, up to £500,000 per day ([TSO, 2003 #510]). All interviewees were in agreement that the targets were non-negotiable and certainty the priority criterion.

According to the interviewees, *cost-effectiveness* emerged as the other priority criterion in the process, since the expected cost of meeting the landfill targets was so high. In the first compliance cost assessment in 1997, the additional cost of meeting the landfill targets in the UK had been estimated at £100-500 million per year by 2010, including an initial capital outlay of £3-7 billion (DETR 1997). The cost saving potential of a TPS

[68] The criteria defined in this report were: degree of certainty provided in terms of meeting targets, practicality/ease of enforcement, environmental benefit, geographical coverage, transparency, costs, flexibility, fairness, and local accountability.

instrument was used to motivate its selection in 2000 (see DETR 2000a p. 33; DETR 2000b p. 87).

Commonly, criteria relating to *equity and fairness*, i.e. who will be winners and losers of an instrument, are important in EPI choice processes (see e.g. Pearson 1995; OECD 2006). However, such criteria do not figure prominently in the official LATS documentation, except for the discussions on how to allocate allowances between local authorities (see **Box 1**). The analysis of political and institutional factors below will explore why such criteria were relatively absent.

Overall, the policy objective and criteria were thus sufficiently clear to allow a systematic assessment of alternative EPIs. Given the two priority choice criteria, a TPS with a defined total level of allowances was a natural candidate for the landfill targets. TPSs provide for certain outcomes by specifying a total level of permits/allowances, and in the case of the LATS this meant that the risk of high fines from the European Court of Justice for exceeding the target could be minimised. TPSs also facilitate economically efficient outcomes by allowing parties to trade permits. In the case of the LATS, this meant that the different cost structures of waste management options in different local authorities could be used so as to minimise the total cost for England in meeting the landfill targets. Eventually, the choice of a TPS instrument was indeed motivated in these terms, i.e. "avoid paying fines", "achiev[ing] the targets in the most cost effective and efficient way", and "giv[ing] those involved the most flexibility in achieving the targets" ([TSO, 2003 #510], para 46-50).

7.4.3 Alternative policy instruments identified

While the initial deliberation on possible instruments by stakeholders beginning in 1997 (see HoL Select Committee on European Communities 1998a) had led to an early interest in a TPS, several alternative EPIs were still considered in the second phase of the LATS process (1998-2000). Whether this resulted from a conscious quest for NEPIs is difficult to assess, since few interviewees were directly involved during this phase and relevant written material from this time is no longer archived by DEFRA. Nevertheless, based on the existing case study material, the EPIs that are theoretically possible can be contrasted with those actually considered (and documented).

Policy instruments for limiting landfill and diverting waste to other waste treatment options could target at least three different stages in the waste management chain (see EEA 2002; DEFRA 2005b p. 5):

- better segregation and sorting of waste to make more efficient use of existing incineration, biological treatment and recycling capacity;
- increased capacity and/or decreased prices of alternative treatment methods;
- and/or increased demand for recyclates and derived products, stimulating expansion and increased competitiveness of alternative treatment methods.

In addition, a fourth stage is the generation of waste. Policy instruments to achieve the BMW targets could thus also be targeted at:

- decreasing waste generation.

Organised around these four stages of the waste management process, a 'long-list' of possible policy instruments at the central government level can be compiled. In **Table 38** below, the instruments listed under each stage have also been classified into EPI types. The list of instruments has been compiled based on five sources: the options considered in the first consultation paper, *Limiting Landfill* (DETR 1999a) (instruments listed in **bold**); the options considered in the ECOTEC consultant report preceding the consultation paper, some as main alternatives and some as supplementary measures (ECOTEC 1999)[69] (instruments listed in *italic*); the European Commission's follow-up review of national BMW strategies and instruments actually used in the Member States (Commission of the European Communities 2005b); a similar EEA review from 2002 (EEA 2002); and a few instruments deduced by the author (in particular in stage 4). Note that the purpose here is not to describe individual instruments, but to *illustrate* the large variety of possible EPIs.

[69] Two proposed supplementary measures in the ECOTEC report are not included here. First, an incineration tax (p. 49) does not stimulate increased diversion from landfill *per se* but has an effect on which treatment method would be chosen for diversion. It could have a minor general effect on waste minimisation, though. Second, the cessation of local authority collection of trade waste (including BMW) (p. 53) would result in an evasion effect (since this waste would then not be defined as 'municipal') rather than genuine diversion.

Table 38. List of possible instruments for achieving the BMW targets

TYPE OF POLICY INSTRUMENT	POSSIBLE INSTRUMENTS AT CENTRAL GOVERNMENT LEVEL (incl. scope and target group)
Stage 1 – Generation of BMW	
Command-and-control regulation	• General legal obligation on waste producers to firstly prevent waste and secondly recover material or energy from waste (Germany) • *Limiting access for mail distribution, through restricting distribution rights*
Economic instruments	• *Provisions for weight- or volume-based collection charges for householders** (several European states) • *Subsidised home composting bins* • *Tax/charge on paper advertising space* • *Raw materials tax*
Liability & damage compensation	• Producer responsibility for BMW streams (paper, newspaper) leading to less resource-intensive and improved design of products**
Education & information	• Public education about waste prevention • Consumer awareness campaigns on 'green' shopping • Information about home composting • Pilot projects on home composting
Voluntary approaches	• Targets for BMW generation
Management & planning	
Stage 2 – Presentation, collection and transfer of BMW	
Command-and-control regulation	• Legal requirement on householders and producers to present separated waste ('presentation by-laws') • Legal requirement for separate collection by municipalities, through kerbside collection, bring banks, or recycling centres (Austria, Germany)
Economic instruments	• *Provisions for weight- or volume-based collection charges for specified streams for householders* (several European states) • *Payment for civic amenity waste/bulk collection* • Grants for municipal separate collection projects • *Charge/refund scheme for newsprint*
Liability & damage compensation	• Producer responsibility for BMW streams (paper, newspaper) requiring producers to provide separate collection facilities
Education & information	• Public education about waste separation
Voluntary approaches	• Targets for separate collection rates of BMW streams (the Netherlands, Portugal)
Management & planning	• Requirement on regions and municipalities to establish waste management plans that address separate collection
Stage 3 – Treatment of BMW	
Command-and-control regulation	• *Landfill ban on all MW* • *Landfill ban on all BW (municipal and other)* • ***Landfill ban on all BMW*** (Sweden) • ***Landfill ban on BMW fractions*** - on all wastes 'not suitable for incineration' (Denmark), 'combustible wastes' (Sweden) or wastes that 'could be recovered because of their nature, quantity and homogeneity' (Flemish region) - on unsorted household and commercial waste (Flemish region) - on waste collected for recovery (Flemish region) - *on waste below a certain TOC* (total organic content)*** level (Austria, Germany, France) - on waste wood (Germany) - on separately collected vegetable, fruit and garden waste (the Netherlands) - ***on paper and putrescibles*** • *Landfill ban on BW fractions* (from municipal and other sources) • Restrictions on landfill of MW • Restrictions on landfill of BMW - bans with annual exemption licences or derogations (Flemish region, Sweden) - ***tonnage permits for landfill site operators to accept BMW*** - ***tonnage permits for waste disposal authorities to send BMW to landfill*** • Ban or restrictions on BMW in specified landfills, e.g. differentiated on urban/rural basis • *Statutory recycling and composting targets for municipalities*
Economic instruments	• ***Landfill tax (incl, increased tax rate)*** • Financial commitment by landfill operators for aftercare (Italy) • ***Tradable permits for landfill of BMW*** - ***for landfill site operators to accept BMW*** - ***for waste disposal authorities to send BMW to landfill*** • *Grants for BMW diversion (per tonne) based on fixed price* • *Grants for BMW diversion (per tonne) through a tendering scheme*

	• *Capital grants for investment in non-landfill waste treatment facilities* • *Reduced VAT on recycling activities* • *Centrally paid supplement to recycling credits*
Liability & damage compensation	• *Producer responsibility: legally binding recycling targets on paper and board, newsprint, textiles, nappies*
Education & information	• Information about non-landfill treatment technologies to businesses and municipalities
Voluntary approaches	• Targets for the use of different waste treatment technologies generally, nationally or per municipality (Walloon region, Sweden) • Targets for recycling/composting of specific BMW streams - food waste (Sweden)
Management & planning	• Requirement on regions and municipalities to establish waste management plans that specify future use of different waste treatment technologies and action strategies (France, Greece)
Stage 4 – Final destination, end use and markets	
Command-and-control regulation	• Legal standards for the quality of recovered material
Economic instruments	• Public procurement favouring products with recovered material input
Liability & damage compensation	
Education & information	• Market research and information (Ireland)
Voluntary approaches	• Industry standards for the quality of recovered material
Management & planning	

*Waste collection charges are normally set locally, but the right to charge and the principles for doing so are often regulated in national legislation.
**Note the 'producer responsibility' is a policy concept that can be implemented either on a voluntary basis or a legal basis. The OECD classifies 'extended producer responsibility' as a liability instrument.
***Note that landfill bans can be formulated as mandatory pre-treatment of waste until it reaches a specified TOC level.
Sources: After ECOTEC (1999), DETR (1999a), EEA (2002) and Commission of the European Communities (2005b).

Although a TPS was a favoured instrument option from an early stage (see DETR 1998), the list demonstrates that there was an early awareness of a very large number of potential instruments. Out of around the 60 instruments in **Table 38**, as many as 25 instruments were identified and documented in the 1999 consultant report (ECOTEC 1999). In the first consultation paper in 1999, these 25 options were narrowed down to five (or seven really) (DETR 1999a):

- Ban on the landfill of all BMW
- Ban on the landfill of certain BMW (paper and card waste, putrescible waste)
- Permits restricting the amount of BMW to be accepted by landfill operators – with the option of making these permits tradeable
- Permits restricting the amount of BMW to be sent to landfill by WDAs – with the option of making these permits tradeable
- (Increasing) the landfill tax.

This is also a relatively high number of alternatives, suggesting that this element of procedural rationality was achieved in the LATS process. Although consultant studies into the design of a TPS for WDAs had already been commissioned, the detailed description of the other instrument alternatives and transparent and systematic comparison suggest that they were more than mere 'pseudo-alternatives'. Furthermore, none of the interviewees expressed disappointment over this stage of the process.

How was the 'long-list' narrowed down to a 'short-list', and eventually the choice of a TPS? Looking at which EPIs were proposed in the ECOTEC report and the 1999 consultation paper, it is clear that most of them can be found under stage 3, i.e. the *waste treatment phase*. This is consistent with the certainty criterion (see previous section), since the landfill targets are *directly targeted* under stage 3, rather than from upstream or downstream in the waste management chain. It should be stressed that most of the instruments in **Table 38** are not mutually exclusive, but would indeed be complementary. Some of them were already in use in the England when the Directive was adopted, and some have been introduced after the LATS process was initiated (Commission of the European Communities 2005b, p. 12).

The certainty criterion is also reflected in the *EPI categories* of actually identified options. The 1999 consultation paper only identified command-and-control regulations (bans and permits) and economic instruments (tradeable permits and landfill tax). Being the two most coercive categories of EPIs (see chapter 2), command-and-control and economic instruments could more strongly predict and guarantee (through the use of legal or financial sanctions) an outcome than for example information measures or a voluntary agreement. Adding the criterion of *cost-effectiveness*, the choice of instrument was furthered narrowed down to a TPS or the landfill tax. Both these instruments can provide a least-cost solution in line with the equimarginal principle, but a tax cannot guarantee a certain outcome, as required by the landfill targets.

To sum up, the LATS process was characterised by documented consideration of a large number of potential EPIs, which contributed to procedural rationality. Even though there was early interest in a particular EPI (a TPS), policy-makers still considered other alternatives in considerable depth. As described above, given the policy objectives and choice criteria, the logic of choosing a TPS is persuasive. As will be seen below, most consultees agreed with this logic too.

7.4.4 Systematic Regulatory Impact Assessment

The immediate purpose of a RIA is to assess impacts of policy instrument alternatives, but it can also have an important function in forcing explicit consideration and documentation of the problem definition, policy objectives, choice criteria, and instrument alternatives identified. Depending on its format and role, it can thus be a key vehicle for the overall procedural rationality of the process. In the LATS process, the seven RIAs performed indeed served as such vehicles by adhering to the standard UK RIA format that addresses all the stages above (see Regulatory Impact Unit 2003). Three of these RIAs were really Compliance Cost Assessments (CCAs)[70] estimating the cost of meeting the landfill targets, including investment in new waste management facilities (DETR 1997; DETR 1998; DETR 2000b). The other four were more specifically concerned with assessing instruments for limiting landfill and were attached to the first consultation paper (DETR 1999a), the Waste and Emissions Trading Bill ([TSO, 2003 #510]), and the draft and final LATS Regulations ([DEFRA, 2003 #598; TSO, 2004 #511]). The information in these RIAs was recycled, though, and not all of them presented a new assessment.

The number of RIAs performed and the fact that they followed the standard format suggest that also this element of procedural rationality was characteristic of the LATS process. They facilitated a more systematic and transparent approach to considering EPIs. However, both the interviews and written material by external stakeholders suggest that there were some issues around the quality of the RIAs.

As mentioned, the first CCA of the landfill targets in 1997 estimated that additional waste management costs in the UK would range from £100-530 million per year in 2010, with initial capital outlays of £3-9 billion (DETR 1997). The 2000 Waste Strategy then modelled a variety of waste management mixes, and concluded that total cumulative costs over the period 2000-2020 of landfill diversion would range from £15-30 billion in present value (DETR 2000b, p. 193). While the EA, local government and landfill industry did not dispute these figures, other stakeholders pointed out a number of conceptual flaws. Unsurprisingly, the incineration industry argued that

> "the CCA significantly overstates the cost of meeting the Directive's targets" (EfWA, in HoL Select Committee on European Communities 1998a, p. 156).

English Nature stated that the assessment omitted the environmental benefit of reducing landfill from the calculation and that effects on reduced waste generation were not considered (HoL Select Committee on European Communities 1998a, p. 25). Friends of

[70] CCAs were the predecessor to RIAs. New RIA guidance was issued in 1998.

the Earth argued that the government overstated the cost by not considering the pre-existing national target to reduce landfill of controlled waste to 60%;

> "it is only acceptable to calculate the additional costs of taking the UK beyond what has already been committed to in Making Waste Work. Unfortunately the Government has not estimated this cost… Therefore the Government's estimated cost of £262-695 million per annum should... be treated with extreme skepticism" (HoL Select Committee on European Communities 1998a, p. 54).

The subsequent four RIAs focusing on *instruments* to reduce landfill were not as criticised. The fact that the costs of diverting waste from landfill were inevitable and a consequence of the landfill targets rather than instruments for implementing them, meant that there were a more limited number of impact parameters on which to compare instruments. The first of these RIAs was a qualitative assessment, which looked at the theoretical advantages and disadvantages with different instrument options only (DETR 1999a). It was followed by the three RIAs attached to the legislative proposals, that compared alternative instrument types and designs although a TPS had already been chosen. Here, impacts such as administrative cost were assessed (around £150,000 per year for the landfill industry and 'marginal' for the local authorities). The costs of some other decisions were also estimated; the cost of including a safety margin (1% would cost £23 million), and the cost saving of delaying the introduction of the scheme by one year (£124 million) (DEFRA 2003c).

However, quantified estimates of two of the most significant impacts are missing in all RIA documents; the *cost savings from making allowances tradable* and *redistributive impacts* between local authorities arising from the allocation. An estimate of the former could have supported the theoretical argument for a TPS as a cost-effective solution (compared to non-tradable permits) and justified the transaction costs associated with trading (i.e. developing a trading strategy and finding buyers/sellers). The DEFRA interviewees said that either they were not aware of any such figures or they did not exist. In the last two RIAs, it was stated that

> "the costs of preventing trading cannot be estimated as we do not have a complete knowledge of all WDAs cost curves for diversion" (DEFRA 2003c, p. 26).

However, the 2000 Waste Strategy (DETR 2000b, p. 87) indicates that such modelling had indeed been undertaken, most likely within the work performed by Enviros consultants. The reason why this work was not referred to more in the public documents was 'commercial sensitivity'. A similar analysis of the Scottish landfill allowance scheme by the same consultant, however, found that the savings from trading would be in the order of 10-15% (about £20-30 million per year in Scotland) (Enviros Aspinwall 2000). Barrow (2003) has estimated that compliance cost could decrease by up to 50% in the initial years by allowing trading, but declining over time. Although most interviewees did not perceive the lack of such a figure as a problem in the LATS process, it is arguably relevant considering the increasing critique from local government towards the end of the process that they would not be able to trade intelligently and thus make full use of the LATS due to various reasons that will be discussed below.

Neither were redistributive impacts arising from the allocation of allowances addressed or quantified in the RIAs. Under the 'competitiveness' sections, only impacts on businesses were addressed, and they were found to not differ greatly depending on the type and design of the instrument. The competitiveness of local authorities, or rather their ability to meet budgets and not be forced to increase council taxes, was not addressed at all. Neither was a study presented of how the allocation of permits would result in winners and losers among local authorities. Despite the absence of such a discussion in the formal documents, however, the equity criterion (see above) and redistributive impacts did figure in the broader political and institutional context, as will be seen below.

In summary, the general UK procedure of RIA helped, if not the identification, at least the documentation of alternative EPIs. The transparent and systematic approach meant that the climate for new EPI ideas could be more open, but as described earlier there was strong agreement among stakeholders that the TPS model identified early on was the best instrument. Since the targets had to be implemented, the results of the RIAs were less important than if the targets would not already have had a legal basis and had to be justified. Still, it is a weakness from a procedural rationality perspective that key impacts such as net cost savings from trading and redistribution between local authorities were not addressed.

7.4.5 Wide consultation

Since the problem definition and policy objectives had been externally determined by the EU, DETR/DEFRA could play more of a 'policy mediator' than a 'policy entrepreneur' role where it would have to make a case for public intervention in the first place (cf. Kingdon 1995). Still, ensuring agreement on the EPI choice was important given the magnitude of the change implied by the landfill targets. Several stakeholders (such as ESA, CIWM and CBI) were dissatisfied with the extent and timing of consultation during the EU negotiations on the Landfill Directive (see HoL Select Committee on European Communities 1998a; HoC Select Committee on Environment Food and Rural Affairs 2005b).

The interviewees did not express such dissatisfaction with consultation undertaken during the second phase of the LATS process. As described in the timeline (see **Table 37**), three formal consultation rounds were undertaken (in 1999, 2001 and 2003) that invited stakeholder views on instrument choice and design features. This exceeded the requirement in the government's Code of Practice on Written Consultation to conduct at least one during policy development (see Cabinet Office 2005). In addition to written consultation, several seminars and workshops were held with stakeholders throughout the instrument design process.

Regarding the consultation outcomes and their influence on decisions, the responses to the first consultation paper would have been interesting to study, but were no longer stored by DEFRA. However, the 2000 Waste Strategy that announced the selection of a TPS reported that out of the 203 respondents over 70% had preferred a TPS for WDAs, i.e. the instrument that was eventually chosen (DETR 2000b, p. 87). About 15% preferred a TPS targeted at landfill operators instead. Thus, there were rather few opponents (<15%) of a TPS instrument *per se*. In addition, the support of a TPS for WDAs allegedly came from a variety of actors (*ibid.*). In the second and third consultation rounds, the vast majority of respondents were borough and district councils, but these consultations focused on technical design issues (DEFRA 2004a; DETR unpublished).

The three formal opportunities for giving views on the instrument choice suggest that the process was relatively open and transparent. The government also accepted the outcome of the first consultation. A TPS had been favoured by the government before, but the consultation led to the decision that WDAs should be the permit holders rather than landfill operators.

7.4.6 Use of evidence

Clearly, the LATS was not an evidence-based policy instrument, since there was no predecessor to emulate, either a TPS for landfill or a TPS targeted at local government (DEFRA 2005e). There seems to have been no major government-orchestrated attempts to learn more generally how to effectively limit landfill from experiences abroad, however. Two actors have since compiled their own international overviews on policies for reducing landfill, CIWM and the Green Alliance (Green Alliance 2002; CIWM 2005).

However, evidence in the form of forward-looking economic and technical analyses was fed into the process. In the first phase of the LATS process, a particular battlefield for evidence was life-cycle analysis (LCA) studies of the comparative environmental performance of different waste management options. Conflicting evidence was referred to by the waste management industry and environmentalists, when the merits of shifting waste from landfill to incineration and/or recycling were debated (see HoL Select Committee on European Communities 1998a). However, this debate did not matter so much for the continued LATS process, which was about limiting landfill rather than identifying the appropriate balance between incineration and recycling.

In the second phase of the LATS process, evidence was collected in the form of technology assessments (see McLanaghan 2002; DEFRA 2005b), the ECOTEC report on possible instruments, and the economic modelling studies on a TPS by Enviros consultants[71].

Overall, a detailed quality review of the evidence used in the LATS process is beyond the scope of this case study. The interviewees, however, did not seem to lack more evidence for instrument design, but rather more forward-looking studies and strategies for landfill diversion generally. It was known that the targets had to be met, but there

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[71] Three separate economic studies have been quoted: Enviros (1999) for DETR, *Landfill Capacity Demand and Prices*; Enviros (1999) for DETR, *Explanation of Modelling to Estimate the Economic Benefits of Tradable Landfill Permits to Meet Requirements of the Landfill Directive*; and Enviros Aspinwall (1999) for DETR, *Landfill Permits, with Statistical Annex*.

was still uncertainty after the LATS had entered into force which management options should be preferred and how barriers to new investment should be overcome.

7.4.7 Summary assessment

The purpose of this assessment of procedural rationality in the LATS process was twofold (see section 4.3.1); first, to assess the degree of procedural rationality as an inherently desirable feature of decision-making, and second, to analyse the extent to which it shaped the final EPI choice. With respect to the first purpose, it can be concluded that there was a high degree of procedural rationality in the LATS process. The process took place in a logical and linear sequence of stages, policy objectives and criteria were clear and agreed upon, a wide range of alternative EPIs were considered originally (although there was early interest in a TPS), several RIAs were undertaken in a systematic format (although some questions can be raised regarding their completeness), wide consultation took place and the consultation outcome on preferred instrument type was accepted, and various forms of evidence informed the process (although the quality could not be assessed in detail here). The overall assessment of each element is summarised in **Table 39**.

Table 39. Overall assessment

Element of procedural rationality	Evidence in LATS process
Linear sequence of decision-making stages	The problem definition crystallised before 1997, targets in 1999, criteria in 1999, instrument type by 2000, instrument design by 2005. Possibly due to externally imposed problem definition and objectives that precluded need for national-level agreement. Previous national-level problem recognition and policy target may not have triggered similar instrument.
Clear policy objective and choice criteria	Quantified and timed targets, which facilitated comparison of alternative instruments. Choice criteria were prioritised and justified; certainty in meeting the targets was necessary to avoid fines from the EU and cost-effectiveness was necessary to minimise the huge investment cost laying ahead for the UK.
Several alternative instruments identified	Originally about 25 instruments were considered, and 7 in the first consultation paper. The ‘viable’ range effectively limited by the key criteria. Certainty ruled out all EPI categories but command-and-control regulation, a TPS, and legal producer responsibility. Cost-effectiveness favoured a TPS.
Regulatory Impact Assessment	Several formal RIAs performed, although their quality was lacking in some aspects. However, the RIAs were important vehicles for including and comparing alternative instruments.
Consultation	There were several rounds with many respondents, broad agreement on preferred EPI.
Use of evidence	The instrument itself was not evidence-based, but various sources of information were used in the process, relating to the environmental performance of waste management options and economic models of diversion costs and landfill demand. However, all evidence was not publicly accessible.

In addition, it may be symptomatic of high procedural rationality that the *process* was not seen as issue by the majority of interviewees. While the idea of EU landfill targets raised much debate, the process of instrument choice was characterised as ‘straightforward’ and the choice of a TPS as ‘uncontroversial’. As expressed by one interviewee,

> “It is not the LATS as an instrument that is critical or interesting here, it is how the diversion can take place and the targets be met on time…”

With respect to the second purpose, it is in the case of the LATS possible to deduce a TPS as the most suitable option given the stated policy objectives and choice criteria. It was described above that a TPS combines the *certainty* of outcome normally associated with command-and-control regulation and the *cost-effectiveness* in reducing total abatement cost associated with economic instruments. It thus appears that specifying clear criteria and systematically comparing the performance of different EPIs in relation to these – in line with the procedural rationality ideal – in this case supported the choice of a previously untested NEPI. Both the consulted actors and the interviewees agreed with this logic and rationale of the LATS.

In the case of the LATS, the UK government thus delivered on its commitments to promoting a ‘professional’ policy-making model, that would consider a wide range of EPIs. The quest for so-called NEPIs was also successful, with the process resulting in the first ever TPS targeted at local government. A tentative conclusion is thus that the LATS process appears to be an example of a rather depoliticised instrument choice. Three important facilitating circumstances seem to be relevant to this case, though. First, the UK actively enforces the general requirement for a standardised RIA procedure and format, that incorporate the elements of procedural rationality (see Regulatory Impact Unit 2003). Second, the fact that the problem definition and policy objectives were determined externally meant that the process could progress without lengthy and iterative challenges of the instrument rationale. The UK government and other stakeholders that were sceptical of the landfill targets had to unite against a ‘common enemy’ and to constructively find an acceptable solution. Finally, the high-cost nature of the landfill diversion problem arguably motivated the government to demonstrate a well-informed and well-justified EPI choice.

What political and institutional circumstances and factors allowed the process to be as straightforward as it was perceived? Furthermore, was this EPI choice really as uncontroversial as it appears in the formal documentation, when considering parallel policy debates and events taking place before and after the choice of instrument type (i.e. the first and third phase of the process)? Having focused on the choice of the LATS as a discrete process and a series of formal events, it is necessary to 'zoom out' and consider the wider political and institutional context, in order to understand whether the process was truly straightforward and depoliticised. Below, the macro-, meso- and micro-level framework developed in chapter 4 (see **Table 16**) is used to support the analysis of political and institutional factors. Several of the factors in this framework overlap with each other and are interrelated in complex ways. Nevertheless, some factors will be found to be more central than others for explaining the LATS process.

7.5 Macro-level factors influencing the LATS process

7.5.1 Legal framework

The EU Landfill Directive, which the UK as a Member State was subject to, was a direct cause of national landfill targets and imperative to the LATS. As mentioned in section 7.4.1, the interviewees believed there would not be a similarly forceful instrument without the Directive. The domestic, non-binding 1995 landfill target would not have triggered much action, since it was expected that already existing instruments would lead to landfill diversion of that order (Anon. 1995a).

Not only did the UK government refer to the principle of subsidiarity when dismissing the idea of reduction targets in the Directive negotiations. The Local Authority Waste Disposal Companies Association argued that

> "these proposals are too prescriptive and restrictive, and retain power to Brussels" (HoL Select Committee on European Communities 1998a, p. 167)

and the Environment Agency somewhat confusingly claimed that

> "true subsidiarity would allow Member States to choose how they met the objectives of a directive" (HoL Select Committee on European Communities 1998a, p. 11).

Unsuccessful in the negotiations, it was seen above that the threat of substantial financial sanctions[72] from the European Court of Justice for not meeting the targets led to certainty of outcome becoming a priority criterion when assessing instruments, hence the promotion of a system of landfill permits (tradeable or non-tradeable). According to Barrow (2003, p. 363), while the UK had originally chosen a tax to promote landfill diversion, the decision on a permit system was “a fait accompli by the EU”. Using the landfill tax as the main instrument to address the targets was thus ruled out in the first consultation paper (DETR 1999a, p. 35).

Meanwhile, no apparent national legislation affected the choice of EPI. There were no national constitutional barriers for central government to place the duty to achieve the targets upon local authorities, which was done in the Waste and Emissions Trading Act. Neither were there any constitutional obstacles to introducing a financial penalty for local authorities who exceeded their annual allowances. Such a regime was unprecedented before the LATS instrument, and it led to much political bargaining, as will be seen below.

To conclude, *the EU legal framework – i.e. the Landfill Directive – strongly shaped the choice of a TPS* in this case, both by introducing the instrument raison-d´être (i.e. the landfill targets) in the first place and by leading to ‘certainty of outcome’ becoming a prioritised choice criterion.

7.5.2 Policy style and political culture

As found in chapter 3, *policy style* is not always precisely defined or consistently used in the EPI literature. To the extent that the repertoire of EPIs defines the policy style, the LATS can be seen as an exception to the British policy style since TPSs were not a standard instrument in the late 1990s (see chapter 6)[73]. The original conception of policy style, however, emphasised two dimensions of the ‘standard operating procedures’ of a national government, that would influence policy approaches,

[72] Note that the estimated fine of up to £500,000 per day had a rather loose basis. It was based on the fine of a single Greek landfill site that had then been upscaled (see EA oral evidence to HoC Select Committee on Environmental Audit (2003).
[73] The first TPS in the UK was introduced in 1998, namely the Packaging Recovery Notes. It thus coincided with the search for instruments to implement the landfill targets, also starting in 1998.

irrespective of the sector or type of policy issue (Richardson, Gustafsson et al. 1982; Buller, Lowe et al. 1993); anticipatory/reactive problem-solving and consensus/imposition relationship with stakeholders. Peters (2000) has in a similar vein discussed the role of public management style, which can be legalistic or managerial.

Considering the typical characterisation of British policy style (see section 3.6.2, **Table 9**), the public management style is *managerial* rather than legalistic. According to Peters (2000), this makes complex and discretionary instruments more likely than command-and-control regulation. The LATS is arguably both more complex and discretionary (in terms of allowing local authorities to themselves make decisions on how to use allowances) than what non-tradeable allowances – one of the most important alternative EPIs considered – would be. Thus, this proposition is supported.

Regarding interaction with stakeholders, the British policy style has been described as *consensual, cooperative* and *consultative* (see **Table 9**). According to Andersen (1999, p. 31), this would result in more open framework regulation rather than stringent command-and-control regulations. Again, a TPS is arguably more flexible and open for target groups than a system of non-tradable allowances.

However, two other aspects of policy style seem more helpful for explaining the LATS process, namely the *reactive* and *discretionary* style of British policy-making. It was argued above that the LATS may not have existed today had there not been any externally imposed targets, since they eliminated the need for national agreement on problem definition and target levels. One of the most reactive views on the landfill targets was that expressed in the ESA's response to the first consultation paper;

> "It is simply not prudent to set unrealistic targets and entrench a culture of failure" (ESA 1999, p. 5).

Also looking at the stakeholder concerns raised in the 1997/98 parliamentary inquiry 'Sustainable Landfill' that considered the prospect of EU landfill targets, it is clear that no targets at all, or targets that would have made other EPIs more suitable, were preferred. First, the reactive style is obvious in the defence of current British landfill practice, made by the Local Authority Waste Dispsal Companies Association;

> "[the Directive] stem from the misguided premise that landfill is bad… It is not true… It must be and is, environmentally benign when properly done" (HoL Select Committee on European Communities 1998a, p. 162).

The DETR, EA and ESA all agreed to this view that UK landfill practice was appropriately regulated and managed, with the LGA pointing out that standards were higher in the UK than in many other European countries. This track record was questioned by Friends of the Earth, who reported several incidents of leaching and other environmental impairment.

In addition to defending landfill *per se*, the industry and government stakeholders also argued that

> "it is debatable if landfill or incineration has least environmental effects and risk to human health" (LGA in HoL Select Committee on European Communities 1998a, p. 171).

Assuming that the targets would lead to a 'dash to incineration', conflicting evidence was referred as to the comparative environmental performance of waste management options. However, while Friends of the Earth were also concerned about increased incineration, they argued that the answer was not to stick to landfill, but to take more drastic action to promote recycling.

Having to accept the EU problem definition of landfill as a greenhouse gas emissions problem, the government and industry stakeholders (including DETR, EA, CIWM and ESA) then advocated an 'end-of-pipe' philosophy by arguing that some form of emission limits would be more appropriate than waste reduction targets. Some quotes illustrate this reactive policy style:

> "It would be more sensible to focus on environmental harm that the Directive seeks to avoid, i.e. methane emission, in line with current United Kingdom regulatory practice" (DETR in HoL Select Committee on European Communities 1998a, p. 64)
>
> "Within the UK we have established a practice whereby we seek to address the need to control emissions through controlling what comes out of the site" (ESA in HoL Select Committee on European Communities 1998a, p. 105)
>
> "Surely a better concept would be to control the gas emissions from the site. It appears that the fundamental starting point of the document is wrongly placed by putting controls on the input element to solve the output problem" (CIWM in HoL Select Committee on European Communities 1998a, p. 37).

These attitudes also revealed that the EU waste hierarchy was not fully accepted as a principle for waste policy. For example, CIWM argued that

> "The Landfill Directive should focus on setting common environmental protection standards for landfills, it should not be used as an instrument to determine a choice between different waste management options" (HoL Select Committee on European Communities 1998a, p. 37).

As a consequence of this 'end-of-pipe' policy style, the technical solutions advocated by these stakeholders were the so-called 'flushing bioreactor' concept, which was a technology for stabilising landfills, and installations for landfill gas capture. The former was disqualified by both the European Commission and the Friends of the Earth as an effective method;

> “”flushing bioreactors” is little more than a last gasp by a landfill industry unable to adapt to changing circumstances and unwilling to accept the challenges of sustainable development” (Friends of the Earth in HoL Select Committee on European Communities 1998a, p. 52).

Regarding increased landfill gas capture, the EA estimated that up to 90% of landfill gas could be captured. At the time (1997), 160 landfills out of about the 500 receiving significant amounts of biodegradable waste had systems for the capture of landfill gas (HoL Select Committee on European Communities 1998a, p. 2). Again strongly questioned by Friends of the Earth and the European Commission, the EA argued that

> “we could certainly make a sound scientific case that we could recover as much methane to deliver the objectives by installing modern gas collection systems” (HoL Select Committee on European Communities 1998a, p. 18).

The government and industry thus wished to redefine the problem from a landfill input problem to an emission problem, which would justify ‘end-of-pipe’ technical solutions. Importantly, an emissions-focused approach would have made a range of other instruments possible, aiming at increasing the efficiency of landfill gas capture and utilisation. Both the waste management industry and professional association preferred traditional command-and-control regulation:

> “imposition of high standards of landfill permitting and operation is welcome in principle” (ESA in HoL Select Committee on European Communities 1998a, p. 85)
> “An alternative regulatory approach would have been to prescribe regulations that all new landfill sites should not emit methane to the atmosphere above a defined limit” (CIWM in HoL Select Committee on European Communities 1998a, p. 40).

Of course, economic instruments such as an emission tax (i.e. using the existing landfill tax which had a rate based on the negative externalities) or tradable emission permits would be possible.

However, both the government and industry argued that traditional regulation would have been more appropriate, since such standards could be considered within the existing practice of applying the Best Practicable Environmental Option (BPEO) in the planning permission process. Importantly, the BPEO maintained the degree of discretion associated with British policy style. The DETR argued that

> “the Directive should not reduce our ability to adopt an environmentally sensible approach to waste management in each case based on current technology and risk assessment” (HoL Select Committee on European Communities 1998a, p. 64).

The need for case-by-case approach to waste management rather than a uniform application of the waste management hierarchy principle was further emphasised by the EA.

> "the waste hierarchy is an important guiding principle to test for sustainability, but is not to be regarded as a strait-jacket, rather each situation must be judged on its own merits" (HoL Select Committee on European Communities 1998a, p. 2).

Considering the propositions stated in **Table 16**, the case study evidence suggests that the choice of LATS was consistent with the managerial and consultative policy style in the UK, if not explained by this style. The choice was inconsistent with the traditionally reactive and discretionary style, on the other hand. Business-as-usual by relying on the BPEO principle and command-and-control regulation on a case-by-case basis was originally preferred. A traditional planning rather than market solution was advocated by industry and government (although an increase of the landfill tax rate was also discussed at the early stages). The LATS thus resulted from an initially involuntary *break with the traditionally reactive and discretionary policy style*. Finally, if one includes the standard EPI repertoire as part of the national policy style, the LATS also represented a break since TPSs or even economic instruments were still relatively rare in the UK in the mid- to late-1990s (see chapter 6).

Closely linked to policy style, the concept of *political culture* has also been used when theorising instrument choice. Linder and Peters (1989 p. 49) have proposed that '*statist*' countries accept more intrusive and centralised instruments. The authors did not define what is meant by statist, but presumably the UK is less statist than for example Sweden[74]. However, the LATS can be seen as a rather intrusive EPI in that quantified input targets are imposed in the (semi-)private market of waste management. For example, it is arguably more intrusive than a tax that does not set definite targets. On the other hand, the trading element increases flexibility for target groups, which may reduce the intrusiveness. As stated by one interviewee,

> "As it became clear that some kind of permits [for landfill of BMW] would be used, the waste management sector saw them as completely changing the commercial conditions… I guess the fact that the permits can be traded gives some freedom and flexibility for the local authorities, but they are still permits, that will be very tough to meet for many".

[74] The authors quote Germany and Scandinavian countries as statist countries. Using the share of government final consumption expenditure of total GDP as an indicator, the UK is less statist than Sweden; government expenditure was 21% of GDP in the UK and 28% of GDP in Sweden in 2005 (World Bank 2006, p. 295 table 4).

Furthermore, it may also be seen as less intrusive than an instrument targeting private actors rather than local government. Lacking a clearer definition of 'statism', however, the tentative conclusion is that the LATS seems to break with the comparatively 'non-statist' political culture in the UK.

Another proposition made in relation to political culture that can be derived from the environmental policy literature, is that a *corporatist political culture* is more conducive to *ecological modernisation*, which in turn is associated with the promotion and use of NEPIs (see Lundqvist 2000 p. 22; Dryzek 1997 p. 151). In the case of the LATS, a NEPI – that was in addition to being an economic instrument also the first TPS ever to be applied to a government body – was chosen despite the UK not being seen as a corporatist country or a historically enthusiastic supporter of the ecological modernisation agenda (Weale 1992; Hajer 1995). This kind of explanation is thus not helpful for understanding the LATS process, but it also needs further theoretical development. For example, is ecological modernisation a necessary and sufficient cause for the adoption of NEPIs? Is ecological modernisation best understood as a concept related to political culture concept or as a party-political and ideological project?

7.5.3 Party politics and ideology

So far we have seen that a precondition for the innovative LATS instrument was the EU legal framework, and that the LATS represented a break with the traditionally reactive and discretionary British policy style. Did this innovative turn have any ideological or party-political roots? A first observation is that there were no signs of dividing lines between the Conservative (-1997) and Labour (1997-) governments on the issue of the EU landfill targets and the selection of a TPS. In the first phase of the LATS process, i.e. the EU negotiations on the nature and level of targets, the Conservative government had disputed the rationale, referring the subsidiarity principle and the overly stringent interpretation of the waste hierarchy. The incoming Labour government continued on this negotiation line from 1997 onwards (Anon. 1997c; Anon. 1997d). The two parties had a common enemy in this phase, namely the European Commission and the proactive Member States, and the key concern in the ensuing EPI choice process was to 'minimise the damage'. The subsequent passing of the Waste and Emissions Trading Bill in the parliament went smoothly (Anon. 2002; Anon. 2003c; Anon. 2003b). Thus, the LATS does not appear to have been a high politics issue or a component of some overarching party-political tactics.

The propositions related to political ideology and EPI choice reviewed in chapter 3 and summarised in **Table 16** are complex to analyse in the case of the LATS. The discussion below will focus on the role of ideology for choosing economic instruments, rather than NEPIs more broadly. Yet, applying the propositions developed in other studies is made difficult by the fact that some pertain to taxes rather than TPSs. Furthermore, some do not consider the different rationales of economic instruments: application of the polluter-pays principle, the facilitation of economically efficient pollution abatement solution, the provision of dynamic incentives, and revenue-raising potential. Nevertheless, we can conclude that there was no Green Party that lobbied for economic instruments (see **Table 16**). Neither was it possible for the Labour government to introduce a tax rather than a TPS due to the certainty criterion (considering that left-of-centre governments have been found to be more likely to support green taxation).

Instead, the relevant question here is whether the LATS resulted from a conscious New Labour agenda of introducing more economic instruments. As described in chapter 2, the Labour government issued a statement of intent when it took office in 1997 that more economic instruments would be used in environmental policy, mainly based on an economic efficiency rationale (see HM Treasury 2002, p. 1). Coinciding with the early LATS instrument choice deliberations, it is possible that this intent was transmitted to the DETR policy analysts but none of the interviewees could provide any direct linkages. The DETR Head of Waste Strategy did acknowledge the possibility of an economic instrument in 1997, though;

> "we will need to weigh up the arguments… as to the preferences for either allowing all [landfill] sites to seek to secure what waste they can within the target, or whether you are going to be dirigiste and pronounce on which sites can have what, or whether you going to seek some sort of market mechanism or not" (HoL Select Committee on European Communities 1998a, p. 72).

While the New Labour aspiration to use more economic instruments could have played a role, it should be noted that they were not first with introducing an economic instrument in waste policy. The 1996 landfill tax adopted by the Conservative government was justified in terms of internalisation of environmental externalities and the tax rate set based on net environmental costs. The same justification was used when the Labour government considered a tax on incineration in the early 2000s. It was ultimately rejected since it was found that net environmental costs were insignificant (HM Treasury 2004b). This suggests that there was some degree of 'ideological continuity', at least in the waste field, in the attitudes towards economic instruments and what constituted an acceptable justification. Unlike other left-of-centre governments, New Labour did not embark on a major green tax shift or give legitimacy to the revenue-raising rationale of economic instruments (i.e. a tax revenue or auctioning of tradeable permits).

The stakeholder responses to the TPS idea also suggests that it was not seen as so ideologically charged, in terms of affording rights and obligations or promoting the traditionally neoliberal value of economic efficiency. The manufacturing industry, through CBI, and the waste management industry, through ESA, welcomed the idea of a TPS (CBI 1999; ESA 1999), although the latter took a pragmatic approach and reserved their position until the details of a TPS scheme had been worked out. The EA also agreed with the TPS proposal, and so did the LGA originally. As will be discussed below, however, the LGA later disputed the economic efficiency rationale. The only ideological opposition found in the case study material came from Friends of the Earth, who argued that

> "the new Bill would introduce a landfill allowance trading scheme which financially rewards councils which incinerate rather than bury their waste" (Friends of the Earth 2003).

However, this opposition related more to the allocation method and the lack of supplementary instruments for stimulating recycling and not incineration, rather than to the principle of or economic theory behind a TPS. Overall, both the written case study material and the interviews suggest that most stakeholders had a pragmatic approach to the selection of EPI and the final choice of a TPS, rather than *a priori* biases or preferences for certain EPIs as had been the case in the 1970s and 1980s (see chapter 1 and 2).

This apparent lack of major ideological arguments around the choice of a TPS is probably related to the fact that the landfill targets were inevitable and set by an external 'common enemy'. As described above, the first LATS consultation paper framed the EPI choice in terms of 'minimising the damage' and ensuring cost-effectiveness, rather than applying the polluter-pays principle or internalising environmental externalities. Indeed, the economic efficiency rationale of a TPS compared with non-tradable permits was rather downplayed through the omission of the cost savings from trading (see section 7.4.4). Speculations were actually made that making permits tradeable might have as much political as economic advantage;

> "intricate central planning fits uneasily into existing relationships between Whitehall and local government" and it could also be "a political minefield generating local protests" (Anon. 1998).

Thus, flexibility for local authorities could have been an informal criterion equally important to cost-effectiveness. It was also noted above that redistributive impacts were not discussed so much at the early stage (DETR 1999a; DEFRA 2003c) (see section 7.4.4), and this may also have served to defuse an ideological debate.

By 2005, the waste management industry viewed economic instruments as both inevitable and essential. The ESA regarded the fact that the UK had taken 'leadership within Europe on the use of economic instruments to deliver environmental outcomes as 'good news' (HoC Select Committee on Environment Food and Rural Affairs 2005b, p. 80). One of the interviewee stated that

> "Government regulation is *the* driver of our sector's future, that is just a fact and we must accept it… Market-based instruments provide incentives for us to continuously improve, in technological terms and also in our pricing… They are just here to stay, really".

The political momentum gradually building up for differentiating waste collection charges for households, is a further testament to the acceptance of economic instruments in the UK.

To conclude, the New Labour interest in using economic instruments may have promoted a TPS solution at the early stage, but on economic efficiency rationale rather than 'polluter-pays' rationale. The economic efficiency rationale was rather weak, however, and dominated by a more pragmatic rationale of increasing the 'flexibility' for local authorities in meeting the targets. Neither was the LATS process a party-political issue, due to the fact that the targets and problem definition and decision to intervene had been made by the EU. Thus, the conclusion here is that the LATS process was not very ideologically charged, hence rather depoliticised. Returning to the propositions derived from the existing literature, it is difficult to determine whether the choice of a TPS resulted from neoliberal ideas of the promotion of economic efficiency or from a left-of-centre agenda of intervening and using harder instruments. This question also depends on what the TPS is compared with – a no-action scenario, using command-and control regulation, or using a tax.

7.6 Meso-level factors influencing the LATS process

7.6.1 Institutional arrangements and organisational culture

The EPI choice literature has emphasised the role of national institutional arrangements (such as formal rules and informal norms, habit and routine, professional backgrounds) and organisational culture as causing path dependency in the choice of instruments and as barriers to adopting NEPIs. There are thus significant overlaps with the concept of national policy style (see above) (see also Linder and Peters 1989, p. 16).

Considering that a NEPI was adopted in the case of the LATS process, the institutional propositions regarding path dependency appear to be invalid. Economic instruments as a group were in the mid-to-late-1990s not institutionalised in the UK EPI repertoire (see chapter 6). TPSs were unusual in Europe and had been experimented with more in the US. Furthermore, the LATS came to be the first ever known TPS to target local government. Thus, the LATS was an innovation that did not fit with the existing institutional framework, which had until then produced predominantly regulatory instruments feeding into the BPEO assessment in the planning permission process (see section 7.5.2 above). First, the landfill targets represented a lack of fit with the traditional waste policy focus within DETR, which was to ensure safety standards for disposal and treatment rather than steer waste streams higher up in the waste hierarchy (see DEFRA Regulation Taskforce 2004). Second, a TPS did not fit with the institutional frameworks at local government level. Local authorities were not used to directly responding to economic incentives and behave as economically rational agents, and thus required new skills and competence (see further discussion below).

How can this break with institutional path dependency be explained then? Institutional arrangements are often contrasted with the influx of new ideas. Although case study material covering the earliest stages of the EPI selection phase (1997-1998) was difficult to access[75], it appears that economists and economic ideas were influential. According to one interviewee,

> "The DEFRA environmental economists were involved from the start, since the costs of the landfill targets were going to be so high for the UK. The aim was to implement them as cost-effectively as possible, this was in the interest of everyone involved… If you look at economic theory, you will see that a TPS can give you a cost-effective solution while also delivering a certain environmental outcome..".

However, the case study evidence does not suggest that there was an 'advocacy coalition' promoting the use of an economic instrument, or a TPS more specifically, mainly because there was no one to advocate against. The advocacy coalition concept would be a more appropriate for analysing the first phase of the process, when the UK stakeholders disagreed with the European Commission and some Member States' idea of landfill targets. With the targets in place, there was a climate of problem-solving to minimise harm rather than advocating competing instrument ideas.

One reason economic ideas were influential was, as stated in the interview quote above, the high-cost nature of the landfill targets. The Treasury insisted that cost-effective solutions should be found to implement the targets (Anon. 1999a; Anon. 1999b). Evidently, this high cost made the institutional investment in adopting a new kind of EPI worthwhile. Another reason could be that the organisational culture within DETR had changed so that the economics profession had become more influential on policy-making in general (cf. Dryzek 1997, p. 102). Linder and Peters (1989) have argued that professional background matters when alternative instruments are analysed. A third reason could of course be that increased influence followed from the New Labour intent to increase the use of economic instruments (see section 7.5.3).

To conclude, institutional arrangements did not act as a barrier to adopting a NEPI in the case of the LATS. However, the catalyst for instrument innovation was arguably that the targets were involuntarily imposed upon the UK. If this had not been the case, it is likely that the traditional style of discretionary regulation would have continued. The innovative choice of EPI may also be a result of professional diversification within DETR and general advocacy of economic ideas.

[75] As mentioned, working documents were no longer archived and key DETR economists had left. I also looked into whether the DEFRA Panel of environmental economists had made statement on the LATS, but was informed they had not.

7.6.2 Policy ideas, learning and transfer

As mentioned above, economic ideas played a role in the LATS becoming adopted, although there was no need for a concerted advocacy to persuade stakeholders of the comparative advantages of a TPS. Other ideational theories on instrument choice have pointed to the role of broader policy context and international policy transfer and competition.

Regarding the broader policy context, it has been proposed that only during ‘paradigmatic changes’ are new types of instruments considered while under normal periods fine-tuning only occurs (Howlett and Ramesh 1993). As described above, the landfill targets were causing a ‘sea change’ in UK waste management, by requiring new conceptual and technological approaches and implying very high costs. This paradigmatic change, visible in the series of waste strategies in the late 1990s and early 2000s (see section 5.4), would thus have opened up for economic ideas on EPIs and eventually the selection of a TPS.

Considering that the LATS is the first known TPS to target local government and one of the few applied within waste policy, there were no pre-existing international models to copy or learn from. Rather, the UK has now developed a niche for and an expertise in TPSs and could themselves transfer the idea. According to some interviewees, other countries have been interested in the LATS;

> “From being one of the laggards internationally with respect to environmental regulation, I would argue that there are now one or two initiatives where the UK has taken a lead and others could learn from.”

To conclude this section on ideational theories, there was no evidence of a concerted advocacy coalition promoting a TPS solution, since there was early broad agreement. Neither was there any evidence of international policy transfer, with the UK pioneering the TPS instrument instead. The paradigmatic change that waste policy was undergoing most likely contributed to the openness towards new instrument ideas. This paradigmatic change was involuntary at first, but with the 2002 Strategy Unit review a more active and self-critical approach was taken.

7.6.3 Policy network characteristics

As described in chapter 3, Bressers and O'Toole (1998) have put forward a theory on instrument choice based on the properties of the policy network active in a certain policy area, i.e. a stable network where actor relationships are continuously reproduced rather than an issue-based community. This complex theory predicting seven different instrument properties essentially proposes that the more integrated the network is, the less costly and coercive the instrument. A TPS such as the LATS is a relatively coercive and costly instrument compared to, for example, information campaigns and VAs. Does this mean that the policy network was not integrated, or more specifically, incohesive and weakly interconnected?

As mentioned above, the main actors involved in the LATS process were DETR/DEFRA, ESA, LGA and EA. Environmental NGOs were not as actively involved. This network was rather strongly cohesive, considering the unanimous opposition towards the targets (which were only welcomed by environmental NGOs and the incinerator industry association EfWA) and preference for traditional discretionary regulation of waste management through the BPEO principle described above (section 7.5.2). According to Bressers and O'Toole (1998), such cohesion would lead to ‘net provision of resources’ and ‘freedom for target groups to opt out’. However, this was not the case with the LATS, which implied costs for the target actors and was compulsory. The network was furthermore not so weakly interconnected, since several formal consultations and informal seminars were held. Strong interconnectedness is supposedly conducive to ‘bilateral arrangements’ (i.e. when there is ‘horizontal’ mutual adjustment of behaviour rather than ‘vertical’ rule setting or order giving from the government, see *ibid*., p. 224), but the LATS applies uniformly to all participants.

To conclude, this policy network theory does not provide helpful explanations of the LATS process. It is too complex to apply in detail here. Indications are that it does not provide an explanation to why a relatively costly and uniform instrument such as a TPS was chosen, though.

7.7 Micro-level factors influencing the LATS process

7.7.1 Actor resources and interests: introduction

The literature focusing on actors' resources, interests and power and their role in determining EPI choice has predicted that the more powerful target group of an instrument, the less coercive the instrument will be (Macdonald 2001) and the more favourable benefit/cost ratio it will involve for the target actor (Daugbjerg 1999). This literature has also emphasised the need to understand the EPI choice process as a process of bargaining between interest groups, where they try to secure their political and economic interests in the issue.

Looking first at the prediction of EPI choice, it is clear that a TPS is more coercive and more costly than other EPI categories, such as investment grants for landfill diversion or technical advisory services. Does this mean that the target group in the case of the LATS was powerless? Compared to the private sector (which this literature has been concerned with so far), local government has arguably less economic power to be exerted towards central government. While local government is economically dependent on and constitutionally subjugated to central government, a cooperative relationship is preferred in the UK. This power asymmetry could thus explain why local government was forced to accept the LATS instrument despite its coerciveness and costliness and why they did not object strongly at the early stages.

Does this relative lack of power explain why the central government in the end chose to target the TPS at local government (WDAs) rather than landfill operators (many of which were private)? DETR had initially rated the latter solution higher, but stated in the first consultation paper that the two options were 'finely balanced'. As described in section 7.4.5, the vast majority of the respondents preferred WDAs to be permit holders. According to most of the interviewees, there were practical rather than political reasons behind this:

> "It is just more logical that those directly responsible for procuring waste treatment services are in control of the permits and ensure the targets will be met, rather than those providing the service. They will have to deal with smaller future contracts anyway, so there's no benefit for them."
>
> "Whether [the permits] addressed local authorities or landfill operators was not really a very important issue in the big picture, which was the enormous amounts of waste that would have to be diverted [from landfill]"

> "There was a measurement issue… It was better that local authorities got their data on waste flows in order [as a consequence of the reporting requirement in the LATS] than asking landfill sites to start collecting data.."

To describe the LATS instrument choice simply as an outcome of the existing power relationships is misleading, however. The political bargaining between central and local government was not this one-way. Below, the LATS process will be analysed as a more complex process of bargaining between the political and economic interests involved, in each of the three phases of the process (see timeline in section 7.3).

7.7.2 Political and economic interests in the problem definition

Strategic, interest-based behaviour can be observed from several stakeholders during the initial phase when the landfill targets emerged. The high-cost nature of the expected landfill diversion meant that various economic interests had to be protected. For example, the underlying interests were rather clear in the critique and defence of the cost estimates presented in the first CCA (see section 7.4.4). The landfill industry and government argued that the total costs would be excessive. For example, the Local Authority Waste Disposal Companies Association, sceptical of the 'arbitrary' targets, asked

> "can Europe afford the cost of these proposals, in order to achieve doubtful and unquantifiable benefits?" (HoL Select Committee on European Communities 1998a p. 167)

The EA agreed that the net cost was unjustified

> "It would place a substantial financial burden on industry, and ultimately the public, to achieve an environmental benefit which is already available through the present UK policy of whole-site collection of gas at landfill sites" (HoL Select Committee on European Communities 1998a, p. 2).

The incineration industry, however, argued that the targets were both 'technically and economically viable' (EfWA in HoL Select Committee on European Communities 1998a, p. 153).

The key reason why the targets were objected to was the feared 'dash to incineration'. Above, the environmental arguments by government and industry for the resistance to such a large-scale shift were quoted (see section 7.5.2). However, there were also political and economic interests at stake. The government had the strong public opinion against incineration to deal with. One interviewee explained that

> "Public attitudes to incineration have been very negative for a very long time, and they keep being reinforced by environmentalists pointing out the health risks and calling for a moratorium… That's the main reason why the government has been so unclear as to how landfill diversion should actually take place, it would be significant political risk"

As for the mainstream waste management industry, there was limited experience with operating modern incinerators and a concern that foreign waste management firms would take market share in the UK. The incineration trade association EfWA, meanwhile, argued that the government wrongly assumed diversion of waste would result in incineration only, rather than a mix of waste treatment options. As opposed to the government's estimate that 160 new incinerators could be needed, the EfWA thought around 35 new incinerators could suffice (HoL Select Committee on European Communities 1998a, pp. 154-156).

As the targets were eventually adopted by the EU, it is interesting to note that government policy towards incineration has since become more cautiously positive. Furthermore, the waste industrial associations ESA and EfWA merged in 2001, reflecting the previously mainstream landfill-dominated industry's adaptation to the new commercial reality. After the adoption of the targets, the waste management industry's critique of the government as being too passive has also increased. The ESA argued in 2005 that

> "compared to many other parts of the EU, the political commitment to become a world leader in recycling and recovery is still lacking and, as a result, the country is continuing to lose economic and environmental opportunities" (HoC Select Committee on Environment Food and Rural Affairs 2005b, p. 85)

It also called for an increased landfill tax rate:

> "There are six major companies in this country that could divert 80% of all landfill waste next week if we had the economic incentives. Because we do not have those economic incentives, it is still cheap to landfill. There is no point in doing it differently. That is why we do not invest in these new processes" (*ibid*. p. 95).

In addition, the government has taken a less critical stance against incineration in the last waste strategies (see chapter 5). Key actors have thus realigned their positions since having to accept the EU problem definition.

7.7.3 Political and economic interests in the choice of EPI type

In the second phase, the principle of a TPS instrument was agreed upon. It has been argued above that it was not a particularly ideologically charged choice of instrument (section 7.5.3). Instead, there was broad stakeholder agreement and a TPS could also be deduced from the priority choice criteria. The political and economic interests were primarily tied to the adoption of the targets *per se*, rather than different instruments for implementing them. Political and economic interests also became more articulated as various design features were worked out (see below). This situation thus corresponds with Daugbjerg's (1999) observation that EPI *types* are not necessarily controversial, but their detailed design can be.

However, it is interesting to note that the waste management industry argued that also 'positive' instruments would be needed, such as capital allowances for recycling plants and relaxed planning and licensing controls (ESA in HoL Select Committee on European Communities 1998a). In addition, they argued that waste strategies had to focus more on waste generation and suggested instruments such as green taxes on virgin material and green procurement, thus shifting the problem from the waste management sector to producers and consumers.

Arguably, the actor with the strongest interest in pushing for a TPS was the DETR itself. A TPS would, beside from reducing the overall costs of landfill diversion, demonstrate a cost-effective and innovative solution to the public, transfer the responsibility of meeting the targets to local authorities, and avoid excessive control of local authorities through the flexibility afforded by trading permits. Furthermore, DETR would not have to monitor the scheme or administer the complicated reporting of waste flows (which the EA would do) or participate in the scheme. In other words, it was easy for DETR to focus on the theory of the instrument rather than its practice.

7.7.4 Political and economic interests in the design of the instrument

Above, it has been described that there was no evidence that local government as the main target group strongly objected to a TPS instrument during the early stage, even though it would impose costs and did not fit with the existing institutional framework. In the third phase of determining design features (2001-2005), however, their critique was stronger and bargaining with central government intensified. Several issues were debated.

First, as mentioned the *allocation of allowances* is a sensitive issue with TPSs, bound to create winners and losers. In the case of the LATS, it seems that the government tried to downplay equity and redistributive issues, by not addressing them explicitly in the RIAs for example (see section 7.4.4). The principle guiding the government (and industry) was that the allocation should reward those authorities that had already made investments in landfill diversion as a consequence of foresight and political leadership, which meant that target year allocations were 'grandfathered' based municipal waste generation (rather than the level of landfill). The LGA was very concerned and stated in a 2004 press release that:

> "Government landfill targets set to be introduced next year could drain millions of pounds from council budgets – leaving town halls with a difficult choice between slashing services or passing costs on to council tax payers… Under the scheme, local authorities who fail to meet Whitehall targets for diverting biodegradable municipal waste such as paper and food from landfill sites will be forced to buy permits from councils who have exceeded their goals – effectively transferring cash from those councils who most need the money to deliver waste services to those which are already on course to meet government targets" (LGA 2004b).

For local authorities that had to buy allowances, the LATS was perceived as little more than a 'huge financial penalty' (HoC Select Committee on Environment Food and Rural Affairs 2005b, p. 69). As mentioned, Friends of the Earth were also sceptical, but on the basis that authorities that had invested in incineration would be rewarded in that annual allowances would actually increase over time leading to windfall profits (Friends of the Earth 2005) (see also section 7.5.3).

An additional sensitivity in this debate was that local authorities with high recycling rates and lower landfill levels tended to be more affluent. In response to the question whether the LATS would hit deprived local authorities harder, the Environment Minister Elliot Morley argued in 2005 that:

> "those local authorities who frankly have been dragging their feet, when faced with the implementation of LATS, have suddenly realised that if they really do not address the issue of waste minimisation, then it is going to cost them and their council taxpayers considerable sums of money" and "it is not so much whether an area is affluent or not affluent, but it is a question of political will from the leadership of the councils" (HoC Select Committee on Environment Food and Rural Affairs 2005b, p. 61).

The allocation method chosen in the end led to windfall profits for about 15 local authorities when the LATS started in April 2005 (see DEFRA 2005f). The 'disproportionate' burden placed on some local authorities became an important argument in the LGA's lobbying for increased funding, however (see below).

Secondly, the LGA strongly opposed the idea to use *financial penalties* for authorities exceeding the allowances. An initial penalty of £200 per tonne BMW had been proposed by DEFRA. Being a new feature for local government, it was rejected on practical grounds. As described by one interviewee;

> "local government in this country is used to implementing statutory targets from central government level. There is no *need* for a penalty. Local government would take responsibility for meeting the allowances anyway, they simply would have to"

A more political concern was voiced by a local councilor and member of the LGA Waste and Environmental Management Executive:

> "I would hope that the government could begin to regard local authorities as true and equal partners. Then I think we will start to make real progress" (Twitchen in Green Alliance 2003, p. 16).

DETR/DEFRA, on the other hand, saw high financial penalties as an integral part of TPSs, essential to enforce the allowance and provide incentives for trading, hence maintaining the credibility of the instrument. The LGA had no understanding for this argument, but were concerned over the effect on local authority budgets and risk-averse behaviour:

> "unrealistic penalty means high costs and 'gold-plating'. The Group remains concerned that the £150 per tonne penalty, wholly unrelated to the cost of diversion from landfill, promotes an extreme safety-first mentality. Most authorities with surpluses will be inclined to bank them for as long as possible, or sell them only at prices close to the penalty figure… LATS penalties should be non-punitive and related to additional costs of diversion" (LGA 2005).

As a response to the strong critique, the government agreed to lower the penalty level from £200 per tonne BMW to £150 per tonne in May 2005[76].

Finally, one of the most serious critiques was the gradually emerging realisation that local authorities may not have the capacity to *participate intelligently in a trading scheme*. The tradable element of the LATS had been welcomed by a large majority of stakeholders when proposed in 1999. However, the cost saving potential offered by trading seemed to decrease as a number of difficulties were pointed out. First, it was argued that there was no real appreciation of the contractual rigidity facing local authorities, preventing a free choice whether to invest in more landfill diversion or buy or borrow allowances.

> "Government believes that landfill trading represents both a stick and an incentive for councils – but with between 5 and 10 year lead-in times for procuring and building new facilities – for many authorities it is simply a stick – an additional and unavoidable cost and in effect a financial penalty" (LGA 2004a)

Second, it emerged that DEFRA and LGA had diametrically different views and expectations on the market for allowances;

> "As the LATS is set up, there has not been a given market price. Local authorities need to know the prices of allowances in the future, so they can budget for them. They don't work like the private sector, where you might accept some uncertainty and can adjust your budget, or go bankrupt for that matter…"

[76] Statutory Instrument 2005 No. 880. The Landfill Allowances and Trading Scheme (England) (Amendment) Regulations 2005.

> "Local authorities must understand that they will *create* the market, that is how a TPS works. It is counter to the whole logic of a TPS to set a price, this should be determined by supply and demand"

The LGA argued that this market uncertainty would lead to risk-averse behaviour, in that councils would not risk tax-payers' money on speculative investment and that they would bank surplus allowances rather than sell them. Furthermore, they argued that the issues around trading drew attention away from the real problem, i.e. to divert waste from landfill and meet the EU targets.

> "There is no reason why landfill trading alone will ensure the delivery of the 2009-10 target, since it simply allows councils to buy off their obligation by purchasing permits from councils which have them to sell without giving a clear picture of the overall amount of new infrastructure (diversion capacity) being built" (LGA in HoC Select Committee on Environment Food and Rural Affairs 2005b, p. 78).

Finally, the new task of developing trading strategies and engaging in transactions was seen as taking valuable staff time and resources from everyday waste management.

In summary, the trading flexibility so strongly promoted at the outset of the process ironically came to be seen as a burden for some local authorities as the market uncertainties and difficulty to behave in an economically rational way as a local government actor became clear. The LGA's response was to demand more 'market intelligence' from DEFRA to inform local authorities LATS planning, i.e. future projections of supply and demand for allowances under different landfill diversion investment scenarios. DEFRA did not see this as their task and it would also be too commercially sensitive to indicate different landfill diversion price levels around the country. This also suggests that the application of an instrument based on economic incentives needs some adaptation in a local government context.

Thus, the economic interests of local authorities became more clear to them as the detailed design was being worked out and implementation drew closer. The analysis above suggests that local authorities would bear the direct costs of the landfill targets, and some of them would be 'losers' within the LATS scheme. We have seen above that this scenario was not just quietly accepted in the third phase of the process. In fact, most of their critique came with calls for dramatically *increased central government funding* of local authority waste management. The negotiations on increased funding did not take place within the immediate LATS process, but in a wider context. This funding can be seen as a compensation towards local authorities for accepting a costly and coercive instruments such as the LATS. Indeed, the funding issue had been seen by both local government and industry as the most crucial to the future of waste management.

Increasing the Local Government Financial Settlement to reflect the costs of landfill diversion, on top of the pre-existing landfill tax, was first addressed in the 2000 Spending Review. DETR wanted more money for waste management to local authorities, but the Treasury disputed the projections, even though they had an agreement on the cost figure for implementing the Landfill Directive (Anon. 1999b). Local government was disappointed with the outcome. One interviewee explained that

> "the government did not seem to realise that costs were escalating rapidly, not just because of the need to divert, but also because waste generation continued to grow quickly… Also, they assumed all of the grant went to waste management, but the reality is that a whole range of other environmental services are financed out of this grant as well.."

After continued lobbying by the LGA, a more comprehensive funding package was presented in the 2004 Spending Review. The Environmental, Protective and Cultural Services Block[77] of the Local Government Financial Settlement, which covered waste management services, was specified as £11.2 billion in 2004/05, gradually increasing to £12.0 billion in 2007/08 (HM Treasury 2004a, p. 117). In 2000/01, the same block had been £8.2 billion (HM Treasury 2000). This meant that the percentage increase in the block from 2000/01 to 2007/08 was almost 50%. However, the LGA was still not satisfied and argued that the real increase was much smaller due to the same reasons stated in the quote above (see also HoC Select Committee on Environment Food and Rural Affairs 2005b, p. 73). Environment Minister Morley disagreed, however:

> "So there is a responsibility on local authorities to be making the necessary investment and to be utilising the money which has gone to them… so there are considerable resources available to local authorities to finance the kind of changes that they need to make to meet these challenges" (HoC Select Committee on Environment Food and Rural Affairs 2005b, p. 56).

Besides the Local Government Financial Settlement, a range of other funding mechanisms were increased in the 2004 package (DEFRA 2004c; HoC Select Committee on Environment Food and Rural Affairs 2005b p. 49). The Waste Performance and Efficiency Grant would contribute £45-110 million per year in 2005-2008. The Waste Minimisation and Recycling Fund would be increased, and additional funding would be made available for Private Finance Initiative (PFI) credits (£150 million by 2007). However, these increases were made at the same time as the

[77] There are eight blocks amongst which local government funding is distributed, with education being the largest followed by the EPCS block.

government expected that efficiency gains of up to £300 million could be made within local authority waste management.

While the increased funding was welcome, several interviewees recognised a more structural problem of underfunding;

> "If you compare the UK to Europe, you will find that we spend only half of what many other countries do on their waste management… We actually spend less than £1 per person each week on waste management. Most people think local authority waste management is much more expensive than it really is, and they think it is a larger part of the council tax bill.."
> "We need to make the economic incentives clearer also at the level of the household. If you produce more waste, you pay more, and vice versa – it's simple. Charges for waste collection should be taken out of the council tax, so that they become visible"

To conclude, it appears that the relative lack of power of local authorities meant that they were faced with a relatively coercive and costly instrument. However, they were also compensated financially 'on the side', which may explain why the LATS process was as smooth as indicated in the assessment of procedural rationality. Furthermore, the section above shows that an analysis of EPI choice based on economic and political interests provides a necessary context for assessing procedural rationality.

7.8 Conclusion

Before drawing conclusions whether the unique and innovative LATS instrument resulted from a depoliticised process characterised by high procedural rationality and why this may have been possible, how has the LATS worked in practice since it entered into force in April 2005? More trading than expected took place in the first year (3% of total allowances) (see **Box 1**). However, it is unknown whether any WDAs exceeded their allowances and were penalised, due to problems and delays in reporting[78]. A National Audit Office (2006a) review argued that effective reporting was required to maintain credibility in the instrument. This review – as well as most interviewees – stated that the key to success for the LATS and landfill targets was to overcome barriers such as limited access to capital for investing in landfill diversion, planning permission delays, negative public attitudes towards incineration, and lack of knowledge of alternative waste treatment technologies. Most stakeholders did not believe that these barriers could be overcome in time to meet the first two EU landfill targets (see HoC

[78] The NAO review issued in July 2006 could not report the outcome of the first year of the LATS. By January 2007, there was still no information on DEFRA's LATS website (http://www.defra.gov.uk/environment/waste/localauth/lats/index.htm).

Select Committee on Environment Food and Rural Affairs 2005b). The NAO estimated that exceeding the 2010 target by 270,000 tonnes and the 2013 target by 1,4 million tonnes could correspond to penalties of up to £40 million and £205 million respectively from the European Court of Justice (National Audit Office 2006a, p. 3).

While the LATS as an instrument may not be successful, the case study evidence suggested that procedural rationality in the process of choosing it as an EPI was successfully ensured. All the six elements of procedural rationality were found to characterise the LATS process to a high degree; a linearly sequenced process, clear policy objectives and criteria, a wide range of alternative EPIs considered, systematic RIA performed, wide consultation, and use of evidence. This result was reinforced by the interviewees' lack of critique of the process of developing the LATS. Furthermore, the results suggest that the LATS choice was not a case of simple *post-hoc* rationalisation, since the predefined two key criteria of certainty of outcome and cost-effectiveness (which were supported by a vast majority of stakeholders and interviewees) were best fulfilled by a TPS solution. Thus, in the case of the LATS process it appears that the UK government has delivered on its commitment to 'professionalise' policy-making generally and to consider alternative EPIs to traditional regulation.

However, it was also argued that the high degree of procedural rationality was facilitated by a set of favourable circumstances. First, the government-wide requirement to undertake RIA according to a standard format was an important vehicle for ensuring all six elements of procedural rationality. Second, the fact that the problem definition and policy objectives were externally imposed meant that the LATS process was more about minimising harm through constructive problem solving rather than re-questioning the underlying instrument rationale. Third, the immediate task here was to limit landfill, which was a more unambiguous and easier task than the broader objective of stimulating landfill diversion. Fourth, the issue at hand was a high-cost problem, which probably increased the incentives to comprehensively search for the most suitable instrument, both to limit the cost and demonstrate to make the eventual decision more acceptable to stakeholders.

To add depth to the analysis of why a TPS was chosen – and why it was so in a procedurally rational way – the macro-, meso-, and micro-level framework of political and institutional factors influencing EPI choice was applied to the case study. These factors overlap in a nested fashion and are not necessarily rival theories but complementary. At the macro level, the fact that this process took place within an EU legal framework was crucial. Had there not been EU landfill targets with credible financial sanctions, the LATS would probably not have existed today, or it might have taken the form of traditional regulation given the existing national policy style at the time. Another contributing macro-level factor seems to be the lack of party-political and ideological battles over the desirability of economic instruments. It was argued that both the Conservative and Labour governments had similar rationales for using them, namely to achieve environmental policy objectives in an efficient way rather than to apply the polluter-pays principle. At the meso-level, it was argued that involving economists to a larger extent in policy-making may have led to a TPS being chosen. Furthermore, the fact that waste policy was undergoing a paradigmatic change appeared to have opened up for new instrument ideas. Finally, at the micro level, the smooth adoption of a rather coercive and costly instruments was possible since local government could not claim their interests towards central government as strongly as private firms may have. Importantly, though, local government were also compensated on the side through increased funding. While these factors all help explain the adoption of a TPS, the analysis also showed that some barriers to adopting NEPIs were overcome. At the macro-level, the reactive and discretionary policy style led to the promotion of traditional command-and-control regulation. The institutional arrangements at the meso-level (which are closely related to policy style) had the same effect.

Overall, an important finding when analysing the political and economic interests of actors and the LATS choice as a bargaining process, was that while the second phase of choice of EPI type was quite straightforward, there was more controversy in first and third phases of determining policy objective and deciding on design features. This suggests that the strong focus on *EPI type* may not be perceived as that important by stakeholders. What matters is how the rationale for public intervention is made and how design features affect the final distribution of costs and benefits and coerciveness.

The high degree of procedural rationality suggests that the LATS process was depoliticised and the choice of a TPS suggests that the NEPI quest was successful. The procedural rationality measures taken suggested that they actively informed the decision, as opposed to merely providing a *post-hoc* rationalisation. However, looking at the whole process and how it ended up in a NEPI also reveals that key political and institutional contextual factors promoted such a choice. Furthermore, the depoliticised choice of EPI type does not mean there were no politics involved in accepting the need for public intervention in the first place or in determining the exact design features.

Chapter 8

The Swedish tax on waste incineration

8.1 Introduction

Like the England waste policy instrument mix, it was found in chapter 6 that the Swedish mix had also been diversified. This begs the question whether EPI choice processes in Sweden have become more conducive to considering alternative and new forms of EPIs. In this chapter, the same analytical framework applied in the previous chapter will be used to analyse the Swedish case study: the *tax on waste incineration*. This tax entered into force on 1 July 2006, almost 15 years after being first proposed and despite strong criticism. As an environmental tax, it is a so-called NEPI. However, as seen in chapter 6, environmental taxes are not really 'new' or innovative EPIs in Sweden, like the LATS was in England. The main purpose of the tax is to stimulate more material recycling and biological treatment[79], by introducing a negative economic incentive for sending household waste to incineration and for investing in expanded incineration capacity. In the bill introducing the tax, other purposes were also specified: to promote more combined heat and power (CHP) generation from waste incineration; to harmonise energy taxation; and to contribute to the green tax shift (Regeringens proposition 2006). It was introduced by incorporating waste as a fuel in the existing energy tax system.

The aim of this chapter is to examine the extent to which the process leading up to the final decision to implement the tax was characterised by procedural rationality. Furthermore, what political and institutional factors influenced the choice of a tax rather than some other type of EPI, and how did these factors influence the opportunities to conduct the process in a procedurally rational way? It will be argued that the process was not characterised by procedural rationality, due to the complexities involved in

[79] Note that the main stimulative effect was expected to be increased materials recycling, rather than biological treatment. In the rest of this chapter, the term 'recycling' will refer to materials recycling only (i.e. not biological treatment or energy recovery).

integrating waste and energy policy, the persistent disagreement among policy actors on the validity of the problem definition and policy objectives, and the early commitment to a tax instrument. The adoption of the tax was mainly determined by current Swedish party politics. In the end, the long choice process led to a compromise outcome that few actors were satisfied with. With an anticipated effect of a 4% increase in recycling, the ‘pro-incineration’ stakeholders argued that the environmental benefit did not justify the costs and the ‘anti-incineration’ stakeholders argued that the recycling incentives it provided were too weak.

The chapter is structured so that the next section gives a brief background to the problem setting and the actors involved (8.2), followed by a time-line of the key decisions and events in the process (section 8.3). Each of the elements of procedural rationality in the process are then analysed in sections 8.4.1 to 8.4.6, with a summary assessment presented in section 8.4.7. Based on these findings, an analysis is made of the role of political and institutional factors at the macro-, meso-, and micro-level in shaping the process of developing the tax (sections 8.5 to 8.7). Finally, section 8.8 draws conclusions as to what may be learnt from this case as an EPI choice process.

8.2 Background

Compared to the LATS instrument, the incineration tax was a domestic initiative. Concerns over dioxin and other emissions had led to a moratorium on incineration in the 1980s, but the public saliency then decreased with the imposition of stricter emission standards (Naturvårdsverket 2002, p. 23). Instead, it was the waste hierarchy principle that inspired interest in limiting incineration and increasing recycling through a tax instrument in the 1990s. By 1995, incineration had become the dominant waste treatment method in Sweden by treating almost 40% of all household waste (see **Figure 6** in chapter 5). However, no decisions were made until there were more acute concerns that an ‘over-capacity’ of incineration was developing in the early 2000s. As a consequence of the landfill bans on organic and combustible waste[80] adopted as a response to the 1999 EU Landfill Directive, an expected shortage in waste management capacity had led to a surge of investments in incineration facilities among local

[80] In 1997, landfill of combustible waste was banned from 2002 and landfill of organic waste was banned from 2005 (Regeringens proposition 1997). Many local authorities were granted short-term exemptions from these bans in order to adapt their waste management systems

authorities. Total incineration capacity was estimated to increase from 3.1 million tonnes waste in 2003 to 5.1 million tonnes in 2008 (Regeringens proposition 2006, p. 65). This rapid expansion was seen as threat to increased recycling. In particular, there were concerns over the incineration of plastic packaging; only 18% of plastics were recycled compared to the 30% producer responsibility target (Statens Offentliga Utredningar 2005a, p. 67).

In addition to the need to replace landfill, it was financially attractive to invest in incineration since it generated incomes from heating and electricity production. About 20% of the national district heating demand in 2005 was met through incineration (RVF 2006c, p. 17). This income made the gate fees for incineration comparatively cheap in a European perspective, and led to an increasing import of waste from neighbouring countries[81] (Statens Offentliga Utredningar 2005a p. 222; Regeringens proposition 2006 p. 35). From a waste policy perspective, there were thus concerns that the rapid expansion of incineration 'crowded out' recycling. From an energy perspective, there was a concern that waste incineration also crowded out biofuel-based energy production with lower CO2 emissions. It was estimated that waste incineration caused about 2% of Sweden's GHG emissions in 2002 (Regeringens proposition 2006, p. 31).

Compared to the problem of limiting landfill such as in the LATS case, policy-makers were here faced with a more complex problem of considering the waste management and energy systems together. This complexity also meant that many actors had a stake in the issue. At the government level, there were some divergent views across the ministries. The *Ministry of Environment* (MoE) was responsible for waste policy and promoted the waste hierarchy, while the *Ministry of Industry, Employment and Communications* (MoI) was responsible for safe and cost-effective energy production and thus had a more positive view on incineration[82]. Since a tax instrument was considered, the *Ministry of Finance* (MoF) led the policy process and their main interest was a legally and administratively feasible solution.

[81] A study found that in 2004 gate fees for incineration were on average 50 EUR per tonne in Sweden and 200 EUR per tonne in Germany. The comparatively cheap prices for incinerating waste in Sweden meant that the import of waste increased from 106,000 tonnes in 1998 to 412,000 tonnes in 2003.

[82] The energy policy unit of the MoI was in 2005 merged with the MoE into a new *Ministry of Sustainable Development*. This did not have a major impact on the energy and waste policy objectives, and the goal conflicts they contained, though. In 2007, this ministry was dissolved.

The *Social Democratic* government had to consider the views of the other six political parties in the Parliament, since they were a minority government in the period 1998-2006. They had a standing cooperative arrangement with – and were dependent upon – the *Green Party* and *Left Party*, which were both keen supporters of an incineration tax[83]. The four right-wing parties (the *Moderates*, the *Liberals*, the *Christian Democrats* and the *Centre Party*) were less anti-incineration and more sceptical of a tax. When the final bill on the tax was laid before parliament in 2006, all of them voted against the tax except for the Liberals.

Some of the party-political disagreement on an incineration tax reflected the divergent views among interest groups. The mainstream waste industry (including local authority-owned waste disposal companies) was represented by the trade association *Swedish Waste Management* (SWM, Swe. Avfall Sverige)[84], who contended that a tax would be unfair and ineffective. They were joined in the opposition by the *Confederation of Swedish Enterprise* (CSE, Swe. Svenskt Näringsliv), since there was a significant level of incineration of industrial waste within some manufacturing industries. They were also joined by the *Swedish Association of Local Authorities* (SALA, Swe. Sveriges Kommuner och Landsting), who were concerned about increased household waste collection charges and the impacts upon local authority investments in incineration as a source of district heating. The *Swedish Recycling Industries' Association* (SRIA, Swe. Återvinningsindustrierna), on the other hand, were positive as their members would gain from the tax. They later allied with the largest environmental NGO, the *Swedish Society for Nature Conservation* (SSNC, Swe. Naturskyddsföreningen), in supporting the tax as a way of implementing the waste hierarchy. In addition to the waste management industry, trade associations representing different energy producers were also involved in the process. Among central government authorities, the *Swedish Environmental Protection Agency* (SEPA, Swe. Naturvårdsverket) shifted opinion during the course of the process and were in the end positive towards a tax. The *Swedish National Tax Board* (SNTB, Swe. Skatteverket), who would administer the tax, had some views on the detailed design of the tax.

[83] In the 2002-2006 parliamentary term, the Social Democrats held 41% of the parliamentary sets, the Left Party had 8% and the Green Party 5%.
[84] Previously, this association was called *Renhållningsverksföreningen*, RVF, in Swedish.

These and other stakeholders and experts were involved in different ways during the policy process, but an important forum for debate was the *government inquiries* that were held (see next section). This is a typical feature of Swedish policy-making, where the government establishes a non-political committee (in this case, one-man committees) and stipulates the inquiry task and premises in a directive (see Statsrådsberedningen 2000). The committee then independently from the government gathers knowledge and evidence and proposes policy measures. The committee is advised by especially invited stakeholder experts throughout its work, although the final proposal is the committee's responsibility (stakeholder members can give 'dissenting opinions'). The inquiry report is then sent out for broad consultation and the responses are collected by the government, who can choose to follow the inquiry proposals or not when it prepares a bill.

The Swedish system of inquiries is often acclaimed for its political independence, knowledge-gathering and consensus-building functions. However, the government can also significantly shape the inquiry outcome through the appointment of the committee members and invitation of stakeholder experts, and through the formulation of the inquiry directive. From a political perspective, establishing a new inquiry can also be a way of postponing an uncomfortable decision. In brief, though, the system of inquiries means that parts of the policy process are normally less within the control by the government than in the UK.

8.3 Key decisions and milestones in the process

Table 40 below provides an overview of the key decisions and publications in relation to the incineration tax. Three phases can be discerned in this process; *initial interest* in a general waste tax in 1990-2001, more *specific investigation* of a separate incineration tax in 2001-2003, and a *final design* of a tax in 2003-2006. This case study will focus on the latter two phases.

Table 40. Timeline of the incineration tax process

Year	Parliamentary and government decisions	Committees of inquiry and other preparatory work
1990		First inquiry report proposing a general waste tax (*Environmental Charges Inquiry*)
1991		
1992		
1993		
1994		Second inquiry report proposing a general waste tax (landfill and incineration) (*Waste Tax Inquiry*)
1995		
1996		Third inquiry report proposing a landfill tax (*Landfill Tax Inquiry*)
1997	Waste policy bill announcing a new landfill tax	
1998		
1999	Landfill tax bill passed in parliament	
2000	January: Landfill tax enters into force	
2001		
2002		SEPA waste report considering an incineration tax Fourth inquiry report evaluating the landfill tax and outlining a proposal for an incineration tax (*2001 Waste Tax Inquiry*)
2003	Waste policy bill announcing a new inquiry to be established for developing more detailed proposal for an incineration tax	Fifth inquiry given directives (*2003 BRAS Inquiry*)
2004		
2005	September: Budget bill announces an incineration tax bill will be laid at the end of 2005	March: Fifth inquiry report proposing an incineration tax (*2003 BRAS Inquiry*)
2006	April: Bill on an incineration tax presented to parliament June: Parliament passes new incineration tax legislation **July: Incineration tax enters into force**	

The consideration of a tax instrument, as opposed to some other EPI, had a long history. The idea was first put on the policy agenda in 1990, when the inquiry on environmental taxes and charges (*Miljöavgiftsutredningen*) identified possible tax bases in different sectors (Statens Offentliga Utredningar 1990). Based on a 1987 SEPA report on waste charging, it outlined a general waste disposal tax, including both landfill and incineration, at an initial rate of SEK 50 per tonne. Four years later, a second inquiry, the Waste Tax Inquiry (*Avfallsskatteutredningen*), was given the task to develop more detailed proposal. It was suggested that a landfill tax only should be adopted, at a rate of SEK 200 per tonne. However, incineration ash sent to landfill should be taxed about four times higher than normal waste, with the effect that incineration and landfill as waste treatment methods would be equally taxed (Statens Offentliga Utredningar 1994, p. 236)[85]. The consultation on the proposals led to the establishment of a new inquiry, since many actors called for a more detailed description and impact assessment.

The directive to this third inquiry, the Landfill Tax Inquiry (*Deponiskatteutredningen*) (Statens Offentliga Utredningar 1996), did not specify a differentiation in rates with respect to incineration ash, meaning that an incineration tax was now off the agenda. This inquiry eventually led to a government bill in 1999 proposing that a landfill tax at 250 SEK per tonne (28 EUR) should be introduced in 2000 (Regeringens proposition 1999). Thus, this first phase led to the adoption of a landfill tax, while the idea to tax incineration was put on hold. An important reason were the landfill bans adopted in 1997 in response to the EU Landfill Directive, which would require expanded incineration to facilitate landfill diversion. Furthermore, modern incineration was seen as a clean and energy-efficient technology by the government. The landfill tax bill did state, however, that the issue of an incineration tax could be reconsidered (*ibid*., p. 24).

Increasing demands for such a reconsideration from the Green Party started the *second phase* of the process in 2001 (see Miljöpartiet de Gröna 2001). They negotiated with the government to start a new inquiry, which resulted in the 2001 Waste Tax Inquiry (*2001 års avfallsskatteutredning*), chaired by a former Green Party MP (Mr. Roy Resare). The purpose of this inquiry was to evaluate the newly introduced landfill tax and analyse a possible incineration tax.

The 2001 Waste Tax Inquiry presented concrete proposals for the design of an incineration tax in its 2002 report. It argued that a weight-based tax on waste sent to incineration at the rate of SEK 400 per tonne (EUR 44) would markedly reduce the incentives to incinerate rather than recycle, through cancelling out the revenues gained from district heating. At the same time it would not be so high that it led to unwanted polluted composts or excessive costs of diversion from incineration. However, it was argued that the tax proposal had to be analysed in more detail before adoption (Statens

[85] In weight terms, about 25% of incinerated waste remains as ash and by-products and must be landfilled. Taxing such incineration by-products at a four times higher rate thus means that the same weight of waste sent to incineration or landfill will be equally taxed.

Offentliga Utredningar 2002). The consultation following this report gave predominantly negative responses (see Ministry of Finance 2002). Still, the Green and Left Parties insisted that a new inquiry analysing an incineration tax should be started. Several of the right-wing opposition parties were critical to a new inquiry, referring to the consultation outcome (Miljö- och Jordbruksutskottet 2002).

The announcement to start a new inquiry in May 2003 (Regeringens proposition 2003), started a *third phase* in the process. The directives for the new inquiry stated that a detailed 'technical-legal design' of an incineration tax should be proposed (Ministry of Finance 2003). It was thus not a case of investigating *whether* a tax should be introduced or not anymore, although its 'suitability' should be assessed. Other economic instruments were also to be considered[86]. The so-called BRAS Inquiry (*BRAS-utredningen*), chaired by a chief judge of an administrative court of appeal (Mr. Curt Rispe), commenced its work in August 2003. After a considerable delay, this inquiry reported in March 2005 (Statens Offentliga Utredningar 2005a). It concluded that an incineration tax was indeed 'suitable', but it proposed that an *energy tax model* rather than the previously considered *weight-based waste tax model* should be used. This meant that the fossil content of waste should be subjected to the existing energy and CO2 tax rates. It was proposed that the revised energy tax legislation could enter into force on 1 January 2007.

Again, the consultation round following this report led to mainly negative responses (Ministry of Finance 2005a). Nevertheless, the government budget negotiations in September 2005 between the Social Democrats, the Left and the Green Party led to a tax according to the energy tax model being announced in the budget bill (Regeringens proposition 2005b, p. 46). Importantly, industrial waste had been excluded from the tax proposal, which now applied to household waste only. The plan to introduce the tax already in January 2006 failed due to delays in the approval from the European Commission on state support grounds and problems with technical measurement methods, and a bill proposing the tax was not issued until March 2006 (Regeringens proposition 2006). After deliberation in the Tax Committee (Skatteutskottet 2006), the

[86] The directive stated that "..[the investigator shall develop a] proposal for how a tax on waste that is incinerated can be designed in legal-technical terms. The committee shall assess the suitability of introducing such a tax or if other economic instruments should be recommended instead" (Ministry of Finance 2003, p. 1).

Parliament decided to adopt the tax on 2 June 2006. The revised energy tax legislation[87] entered into force on 1 July 2006.

By 2005, the final design of the tax was thus more or less clear, after more than a decade of debate. The resulting tax is described in detail in **Box 2**. In brief, the tax applies to the fossil carbon content of household waste (which consists mainly of plastics and rubber products in household waste), which is calculated by a standard proxy of 12.6% of the weight of household waste. As was described above, the tax rate was not set based on estimations of the negative environmental externalities from incineration (see e.g. Pearce and Turner 1990 pp. 84-100). Instead, pre-existing energy tax rates were used, namely 150 SEK per tonne fossil carbon (c. 16 EUR) for the fiscal energy tax and 3,374 SEK per tonne fossil carbon (c. 367 EUR) for the CO2 tax. In order to stimulate CHP production, these tax rates are differentiated so that incinerators producing heating only pay the full tax rate, while CHP incineration plants pay significantly lower rates. About two-thirds of the 30 incinerators subject to the tax have CHP technology installed.

Box 2. Detailed description of the incineration tax

The proposal for a tax instrument to curb incineration evolved from a weight-based tax on the tonnage of waste sent to incineration to an inclusion of waste among the fuels subject to **energy and CO2 taxation**. The **tax base** is waste as a fuel for producing energy (heating and electricity). More specifically, the tax base is delimited to household waste (i.e. waste that is subject to the municipal waste collection duty, as well as imported household waste), and does not include industrial waste as originally proposed. Furthermore, the tax base is delimited to the fossil carbon content of household waste. The law stipulates that the proxy for fossil carbon content to be used is 12.6% by weight of household waste. The **tax point** is the consumption of the fossil carbon, i.e. when the waste is incinerated. The targets of the tax are those commercial facilities that incinerate more than one tonne of fossil carbon in household waste per month, in total about 30 facilities (Regeringens proposition 2006, p. 51).

The energy taxation package contain three different taxes; an energy tax which is purely fiscal, and a sulphur tax and CO2 tax which are both externality-based taxes. It was decided that the sulphur tax should not apply to waste, due to the low sulphur content. The **tax rates** on the other two taxes were set on different rationales (see table below). The CO2 tax rate is calculated for each fuel based on its CO2 emissions, which meant SEK 425 per tonne household waste. The energy tax rate for different fuels, on the other hand, does not follow a similar logic. The government set the energy tax rate at SEK 150 per tonne fossil carbon, i.e. SEK 19 per tonne household waste. It was argued that this low rate, compared to other fossil fuels, was attractive from a national energy supply perspective. However, the rate could not be set too low given the rules on state support which prohibit significant exemptions that act as indirect subsidies (Regeringens proposition 2006, pp. 49-50). These tax rates apply for 2007, but are then due for annual recalculation.

SECTOR	**ENERGY TAX**	**CO2 TAX**	**TOTAL TAX**
Household waste with 12.6% fossil carbon content (by weight), tax SEK per tonne (*in EUR*)[1]			
Heating plant (hot water boiler)	19 (*2*)	425 (*46*)	444 (*48*)
Power plant (condensing power plants)[2]	0	0	0
Combined heat and power (CHP) plant[3]	0	76 (*8*)	76 (*8*)
Manufacturing industry[4]	0	89 (*10*)	89 (*10*)

[87] *Lag (2006:925) om ändring i lagen (1994:1776) om skatt på energi* (Eng. Law (2006:925) on amendment to the law (1994:1776) on energy taxation).

[1] 1 EUR = 9.20 SEK.
[2] Electricity production (no heating) is 100% exempted from energy and CO2 tax.
[3] The taxation of CHP depends on the actual efficiency (the proportion of electricity to total energy) of the plant. For the proportion of the fuel used for electricity production, there is 100% exemption from both energy and CO2 tax. For the proportion of the fuel used for heat production, there is 100% exemption from the energy tax and 19-79% exemption from the CO2 tax when the efficiency is 5% or higher. The maximum exemption from CO2 tax (79%) applies when the efficiency is above 15%, and this is the tax rate shown in the table above.
[4] According to the law on energy taxation, no energy tax and only 21% of the CO2 tax applies for fuels that are used within manufacturing industry and in the agriculture, forestry and water sectors. Note that only marginal amounts of household waste is combusted in manufacturing industry incinerators.
Sources: Regeringens proposition (2006), Statens Offentliga Utredningar (2005a, p. 40).

An important objective of the tax was to incentivise the combined CHP in waste incineration, rather than heating only. Therefore, the tax rates have been **differentiated** for incinerators that produce heating only and those that produce both heating and electricity. Given that about two-thirds of the incinerators for household waste have CHP capacity, this means that most tax-payers will pay lower rates than those listed above, namely zero energy tax and around SEK 76 per tonne household waste in CO2 tax depending on the ratio heating/electricity.

The tax is administered together with other energy and CO2 taxes by the Swedish National Tax Board. Those with a tax duty are required to register with the Board and then **report** on waste tonnages on a monthly basis, unless the total annual tonnage is expected to give rise to a total annual tax of less than SEK 20,000.

The bill stated that the tax should be **evaluated** after its introduction, *inter alia* to assess whether complementary instruments were necessary. It was stated that inclusion of industrial waste may be considered, based on more extensive RIA, and that the CO2 tax may need to be reconsidered depending on the outcome of EU Emssions Trading Scheme negotiations. (Regeringens proposition 2006, p. 44, 56).

Since the tax was only introduced on 1 July 2006, it is too early to evaluate its **effects**. The *ex ante* RIA performed by the inquiry putting forward the final tax proposal estimated that about 4% more of municipal waste would be source separated and diverted to materials recycling (Statens Offentliga Utredningar 2005a, p. 173). It was estimated that the total tax receipt for household waste would be about SEK 479 million per year (p. 183). This increased incineration cost was estimated to lead to increases in the waste collection charges for households of 3-27% (p.183).

Sources: Statens Offentliga Utredningar (2005a), Regeringens proposition (2006).

8.4 Procedural rationality in the incineration tax process

This EPI choice process finally resulting in the tax on incineration thus took place over one and a half decade, with a more intense phase of policy analysis and debate from 2001 onwards. Below, an analysis is made of the extent to which the six elements of procedural rationality (see **Table 15** in chapter 4) characterised the process: linear sequence of decision-making stages, clear policy objective and choice criteria, identification of several alternative instruments, systematic Regulatory Impact Assessment, consultation, and use of evidence.

8.4.1 A linear sequence of decision-making stages

As suggested by the chronology, and in contrast to the LATS case, there were no sequential stages in this process. Instead, the 'stages' were dealt with in parallel to each other and were not concluded, due to the fact that the rationale of the tax was constantly disputed by various stakeholders. The fact that the process took so long was frustrating to both pro- and anti-incineration stakeholders that were interviewed:

> "For investors [in incineration facilities], certainty is a key factor. We would almost have preferred a decision to introduce the tax earlier on, than this prolonged debate and indecisiveness from the government"
> "The process just went on and on because the government did not really want the tax. They kept putting off an uncomfortable decision"

The *problem definition*, which was not externally determined in this case, was still debated by 2005, when the decision was finally made. Moving up the waste hierarchy had been the original rationale. Later in the process, the validity of the waste hierarchy, and in particular the comparative environmental performance of material recycling and incineration, was questioned in a wider waste policy debate (see further discussion in section 8.6.2). On a more practical level, there were conflicting views whether incineration really crowded out recycling, whether there would be an 'over-capacity' of incinerators or not[88], whether recycling should be stimulated in general or just for the plastics waste stream, and whether waste as a fuel would be replaced by biofuels or fossil fuels. The government avoided to make an explicit statement of the problem to be solved in its directives to the 2001 and 2003 inquiries, by just specifying that consequences of a tax should be analysed (Ministry of Finance 2001; Ministry of Finance 2003). Consequently, the inquiries discussed the problem issues comprehensively, but refrained from pinpointing it and asking whether the problem *justified* a policy intervention. Several interviewees who were members of the second inquiry were critical:

> "A proper needs analysis or problem analysis was never made. An open, unbiased analysis was just side-lined… To me, this decision-making style is illogical and irrational"
> "We could never agree within the inquiry committee [on a problem definition], but, on the other hand, this was not really the task either, since the inquiry directives had effectively closed the issue".

Regarding the determination of *policy objectives* and *choice criteria*, these too were disputed up until the decision to introduce the tax in 2005. As will be seen in the next section, objectives, criteria and 'considerations' were added and modified during the course of the process. Again, the inquiry directives were unclear and specified an array of partially conflicting waste, energy and taxation policy objectives, which left the inquiry committees to speculate and reason which should be prioritised. A quantified target for increased material recycling was not set until 2005 (to recycle at least 50% of

[88] In 2000, the Swedish EPA did a survey of plans for landfill diversion as a consequence of the landfill bans and found that there would be a 1 million tonne shortage in waste treatment capacity in 2002, despite increases in recycling and biological treatment capacity (Naturvårdsverket 2000a). A new consultant survey in 2001 found that there were known plans to expand incineration capacity so much that it would double from 2000 to 2006 (Profu 2001, in Statens Offentliga Utredningar 2005 p. 237).

household waste by 2010, see Regeringens proposition 2005a), but it did not influence the choice and design of the tax, according to interviewees.

The chronology in the previous section showed that there was an early (1990) and sustained *a priori* commitment to a tax instrument, meaning that there was no distinct stage of *identifying alternative instruments*. This commitment had historical roots and also political motives, which will be examined in later sections. It meant that the choice of EPI preceded the problem definition and setting of objectives. Several interviewees were annoyed by this '*post-hoc* rationalisation' of a tax as the best tool for the job:

> "Much of the inquiry work was about finding objectives that could be achieved by a tax… It was not an objectives-led process where you are open to alternative solutions"

The conclusion of the 2003 BRAS Inquiry is illustrative:

> "A comprehensive assessment of the [tax] proposal suggests that it is suitable to introduce, above all since it steers towards several relevant policy objectives" (Ministry of Finance 2005b).

Although the *a priori* commitment to a tax was very clear to everyone involved, one interviewee was particularly cynical about the tactical use of an inquiry for this purpose:

> "Government inquiries are supposed to be independent from government and conduct their own analysis of a problem and propose policy responses. In this case, it was purely a question of commissioning someone to look at the legal aspects of a tax, and involving stakeholders in the process to try and get acceptance of the tax before it was introduced".

In summary, this process did not have a linear sequence of stages. The instrument was identified first, followed by protracted debates on problem definition and policy objectives. In addition to the political *a priori* commitment to a tax instrument, some factors related to the problem setting could explain the non-linearity, especially when comparing to the circumstances of the LATS case. First, the fact that incineration is both a waste and energy policy issue, meant that two complex production systems had to be considered in the problem analysis and multiple objectives were inevitable. Second, the problem here was more delicate, namely to fine-tune the balance of incineration and recycling (both preferable to landfill), than to to impose absolute limits on a management option that was at the bottom of the hierarchy. Third, the problem and objectives were not externally imposed upon Swedish policy actors, but the idea of a tax to stimulate recycling had a domestic origin was proactive rather than reactive. There were thus more issues to secure agreement upon among stakeholders, and the lack of clear government policy meant that these debates were given much room.

8.4.2 Clear policy objectives and criteria

A historical review of the early inquiries into a combined waste tax (landfill and incineration) reveals that the main objective evolved from implementing the polluter-pays principle and increasing the use of economic instruments generally (Statens Offentliga Utredningar 1990), to decreasing the waste generation and promoting source separation (Statens Offentliga Utredningar 1994), to reduce the amount of landfilled waste (by a landfill tax only) (Statens Offentliga Utredningar 1996). When an incineration tax was reconsidered in the early 2000s, the first government inquiry directive did not specify an overall objective of a tax (Ministry of Finance 2001) while the second described the overall objective as ‘stimulation of recycling’ (Ministry of Finance 2003). Compared with the landfill targets in the LATS case, this objective was thus *imprecise* (in terms of which waste streams should be diverted to recycling) and *unquantified*. Underlying this vague objective was of course the ambiguities of and controversy around the waste hierarchy. This broad objective meant that a variety of EPIs in addition could be used to address it, something which the stakeholders also suggested (see next section).

In addition to the lack of a clearly stated primary objective, a range of other policy instruments and circumstances that the 2001 inquiry committee should take into account were specified in the directives, such as the landfill tax and bans, the energy tax reform, the new green tax shift programme, the EU incineration directive, and the EU state support rules (Ministry of Finance 2001). By 2003 the list of ‘considerations’ had grown further, to include the green electricity certificates scheme, a possible district heating tax, CO2 emissions trading, and the new food waste recycling targets (Ministry of Finance 2003). In both these inquiries, the investigators thus had a formidable task in clarifying what the relevant objectives and targets really were, and then interpret and prioritise them, if not explicitly then implicitly. Some interviewees argued that this was symptomatic of ‘policy congestion’ (cf. Majone 1989; Sorrell and Sijm 2003) in the waste field:

> “Before we start on a new thing [i.e. the incineration tax] we need to get a better overview of waste policy – how instruments affect each other, what signals are sent, whether the signals are contradictory, how we can use existing instruments better… Right now, it is just too messy and no one has the overview… [The consequence is that] we come up with new instruments to solve the problems created by other instruments”

Others argued that such an overview and coordination task was beyond the responsibility of the inquiries:

> “Yes, it is a complicated context, but we need to simplify and focus on the waste objective here, which is to increase recycling… Interaction effects are just inevitable”

Besides from the vague waste policy objective, there were some *goal conflicts* with Swedish *energy policy objectives* (see Regeringens proposition 2001), which stated that the energy supply should be safe, cost-effective, preferably domestic, and with a low negative environmental impact. Waste incineration offered a safe, durable and domestic supply of energy. Furthermore, the negative price of waste as a fuel meant that incineration was a cost-effective source of energy that would not harm international competitiveness. From an energy policy perspective, waste incineration was thus a good option for energy production. Regarding *climate policy objectives* and CO2 reduction targets, however, waste incineration was a less attractive option insofar as it competed with biofuels with lower CO2 emissions, but more attractive insofar as it competed with fossil fuels.

The objective of the *green tax shift* (see also section 6.2.3) was also mentioned in the inquiry directives, while not stated as an explicit purpose. Continued progress with this shift required the establishment of new environmental tax bases. A potential goal conflict with this objective, is that the green tax shift requires relatively stable tax bases. This means that continued incineration, with due payment of taxes, would be preferable to significant diversion of waste to recycling, as a consequence of the negative economic incentive to incinerate.

As for *choice criteria,* the directive specified generic policy criteria only and did not propose a ranking of their relative importance.

> ”A tax should guarantee a shift towards cost-effective systems and in a holistic perspective the achievement of relevant environmental objectives… In legal-technical terms the solution shall guarantee good function, simple implementation and reasonable control possibilities” (Ministry of Finance 2003).

Equity issues were not explicitly referred to, but it was specified that ‘economic consequences should be considered. Compared to the LATS case, the lack of priority criteria and the somewhat fuzzy overall task to assess a tax’s ‘suitability’ suggests that more discretion was given the inquiry committee in making the assessment.

In light of these vague and conflicting policy objectives and generic criteria, could the choice of a tax, and more specifically the energy tax model, be deduced, like in the case of the LATS? Obviously, the decision to adopt a tax (as opposed to not doing this) meant that waste policy objectives were prioritised over the energy policy objectives, *or* that the green tax shift objective took precedence. Interestingly, though, another objective became increasingly emphasised towards the end of the last inquiry and in the ensuing government bill, namely (the inherent value in) *harmonising the energy tax system*. With such an objective, the chosen instrument was the only possible option.

To summarise, clear policy objectives and criteria as a key element of procedural rationality did not characterise the incineration tax process. The lack of clarity, precision and prioritisation and the existence of difficult goal conflicts would have made a systematic and transparent comparison of alternative instruments – had it taken place – challenging. However, to have a multitude of partially conflicting objectives is probably more common than not in policy-making. More seriously, was that many stakeholders perceived that some objectives served only to provide a *post-hoc* rationalisation of the *a priori* chosen tax instrument.

8.4.3 Alternative policy instruments identified

As described above, there was an *a priori* commitment to a tax and therefore no real stage of comparing EPIs. However, different design choices were considered in the 2001 and 2003 inquiries. In the latter, some complementary economic instruments were also considered. Like in the LATS case study, to illustrate the variety of *possible* instruments for achieving increased recycling (assuming this to be a primary objective) a list has been compiled in **Table 41**. First, the only tax model (although at different rates) proposed in the 2001 inquiry is listed in *italics* (Statens Offentliga Utredningar 2002). Second, the two tax models and three complementary economic instruments identified in the 2003 inquiry are listed in **bold** (Statens Offentliga Utredningar 2005a). Thirdly, the alternative instruments suggested in the consultation responses to these two inquiry reports have been included (Ministry of Finance 2002; Ministry of Finance 2005a). Finally, the five instruments identified in the 1993 waste tax inquiry are listed (Statens Offentliga Utredningar 1994). Beside the categorisation of EPI types, the instruments have been grouped into whether they primarily address reduced or stabilised use of incineration, or the stimulation of material recycling and biological treatment, or other stages in the waste management chain.

Table 41. List of possible instruments for increasing materials recycling

TYPE OF POLICY INSTRUMENT	POSSIBLE INSTRUMENTS AT CENTRAL GOVERNMENT LEVEL
Reduce expansion of incineration	
Command-and-control regulation	• Regulation on which wastes can be sent to incineration (2005 consultation)
Economic instruments	• Waste tax – landfilled and incinerated waste taxed equally (1993 Waste Tax Inquiry) • Waste charge – landfilled and incinerated waste charged equally, payable to an environmental fund instead of treasury (1993 Waste Tax Inquiry) • *Waste tax model* - Weight-based tax on incinerated waste (2001 Waste Tax Inquiry, 2003 BRAS inquiry): - Different tax rates - Differentiated tax rate depending on: CPH efficiency, classification of wastes, ash production, heating efficiency - Exemptions: industrial waste, hazardous waste, sludge, animal by-products, etc. • **Energy tax model** – Incorporation of (fossil carbon content of) waste as a fuel subject to energy and CO2 taxes (2003 BRAS inquiry): - Different tax rates - Differentiated tax rate depending on: CPH efficiency - Exemptions: hazardous waste, industrial waste - Measurement of fossil carbon content: proxy or actual measuring • **Combined waste tax and charge**: general weight-based tax on waste to reduce generation, with specific charges set for each treatment method depending on their environmental performance that are returned to the industry (2003 BRAS inquiry, 2002 and 2005 consultations) • Inclusion of incinerators in CO2 emissions trading system (2005 consultation)
Liability & damage compensation	
Education & information	
Voluntary approaches	
Management & planning	
Increase material recycling and biological treatment	
Command-and-control regulation	
Economic instruments	• **Tradable recycling certificates for plastics** (2003 BRAS inquiry, 2005 consultation) • Subsidies for biological treatment (2002 consultation) • R&D support grants to recycling technologies (2002 and 2005 consultation) • Tax exemption for biogas as a vehicle fuel (2002 consultation)
Liability & damage compensation	
Education & information	• Education and information for waste managers (2002 and 2005 consultation)
Voluntary approaches	• Environmental quality labelling of fertiliser produced from biological treatment of waste (2005 consultation)
Management & planning	
Other	
Command-and-control regulation	• Stricter regulations and enforcement of competition regulation on waste management market (2005 consultation)
Economic instruments	• **Raw materials tax**: stimulate demand for recycled materials (2003 BRAS inquiry, 2005 consultation) • **Product tax**: penalise hazardous substances in products and facilitate recycling (2003 BRAS inquiry, 2002 and 2005 consultations) • Increased landfill tax (2002 and 2005 consultation)
Liability & damage compensation	• Lifecycle guarantee: duty for producer to demonstrate financing of producer responsibility for products with long lifetime (1993 Waste Tax Inquiry) • Producer responsibility: higher recycling targets and better follow-up (2002 and 2005 consultation)
Education & information	• Resource savings committee to disseminate information and provide education and research (1993 Waste Tax Inquiry) • Environmental product declarations (1993 Waste Tax Inquiry) • Information to stimulate demand for recycled products and materials (2002 consultation)
Voluntary approaches	
Management & planning	• Improved collection infrastructure and access to source separation stations for producer responsibility materials (2005 consultation)

Sources: Statens Offentliga Utredningar (1994); Ministry of Finance (2002); Statens Offentliga Utredningar (2002); Ministry of Finance (2005a); Statens Offentliga Utredningar (2005a).

The table above shows that *fewer instruments* were proposed and/or considered than in the LATS process, around 25 on the whole. Most of the instruments considered in the government inquiries were naturally tax-related, but the 1993 inquiry identified also some information and liability EPIs. As mentioned above, the 2003 BRAS Inquiry had to propose a tax, but was also instructed to consider other economic instruments, specifically of 'a more positive character' (Ministry of Finance 2003). The investigator presented two tax instruments that were comprehensively compared, the *energy tax model* and the *weight-based waste tax model*. In addition, three 'complementary' economic instruments were analysed (Statens Offentliga Utredningar 2005a, pp. 305-331):

- a combined waste tax and charge,
- tradable recycling certificates for plastics waste (i.e. a TPS), and
- raw materials and product taxes.

These were seen as 'complementary' rather than 'alternative' after the inquiry committee had decided an energy tax model should be recommended. Also the weight-based waste tax model was also seen as 'complementary' in the end, as it could be introduced on top of the energy tax model. The three complementary instruments were described but not assessed in a quantitative way. All three were seen as interesting for future waste policy, in particular the third, but would require further analysis.

Unsurprisingly, those interviewees that supported the idea of a tax were not concerned over the few alternative EPIs identified:

> "A tax is just the easiest and best way to tackle the incineration problem right now. It is an instrument that everyone is familiar with. The system of tradeable recycling certificates sounds interesting, but… We could consider other instruments, but that can be done at a later stage"

Other interviewees who had opposed the need for a tax argued that

> "too few alternatives were looked at, and not in sufficient detail"
>
> "the three 'complementary measures' or whatever, they were just 'pseudo-alternatives' anyway… The government had thrown in that instruction [to identify other economic instruments] in the directive to make the whole effort look a little more rational and solid. But no one expected a thorough assessment of them, or that they would have an impact on the proposal for an incineration tax"

Meanwhile, the inquiry secretariat felt that the formulation of the directive, and the lack of time and resources, did not allow them to examine alternatives more in-depth.

The majority of EPIs proposed originated instead from the various stakeholders in their consultation responses (see **Table 41**). Partly as a consequence of the vague policy objectives stated in the inquiry report, both 'harder' instruments, like high taxes, and 'softer' instruments, like better information and technology subsidies, were proposed.

Also, the qualitatively formulated objective could potentially be achieved both by broad instruments targeting all waste (like a tax) or more precisely targeted instruments (like a strengthened producer responsibility for plastics). Finally, the formulation of the objective meant that both negative instruments addressing incineration and positive instruments addressing recycling could have been chosen. Regarding the balance between the six categories of EPIs, mostly *economic instruments* were considered, as could be expected by the commitment to a tax. *Command-and-control regulation* specifying which wastes should not be accepted in incinerators were not popular among any of the stakeholders. Among *liability instruments,* more ambitious targets for the producer responsibility in relation to the under-performing plastics waste stream and better enforcement were commonly called for. *Voluntary approaches* were not seen as sufficient for causing a larger shift and investment in new recycling infrastructure. *Education and information* initiatives were proposed by some to inform industries of recycling technologies and opportunities. Finally, more effective source separation, a prerequisite for increased material recycling and biological treatment, was also emphasised as an alternative to a tax throughout the process.

More than in the LATS process, interaction with existing instruments had to be considered due to the many links with both waste and energy policy instruments. For example, it was unclear if the green electricity certificate scheme to promote renewables should include organic waste fractions as renewable fuels, and hence reward and incentivise incineration[89]. It was quite a complex landscape in which a new instrument had to fit in, making it more of a 'second-best' choice than 'first-best' choice. Some stakeholders therefore argued that existing instruments should be tuned up or better enforced, instead of introducing a new one. As explained by one interviewee,

> "I don't understand why they cannot use existing instruments more, and ensure that they work properly or go in and modify them so that lead to better results… In this whole process, [the government] seemed fixated on the idea to introduce a *new* instrument, a tax".

In summary, no real alternative instruments were considered by the 2001 and 2003 inquiries, as a consequence of the narrow directives. Some 'complementary' instruments were identified, but several interviewees saw this only as a *post-hoc*

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[89] The 2003 BRAS Inquiry was given the task to investigate the issue of green electricity certificates and whether it contradicted a possible incineration tax. In recent cases, certain biomass waste fractions had been awarded status as a renewable fuel and thus been financially rewarded by the scheme. The inquiry concluded that the 'renewable' fraction of household waste should not qualify as a renewable energy source eligible for certificates (Statens Offentliga Utredningar 2005b).

rationalisation of the tax choice. Meanwhile, stakeholders proposed a range of different instruments, and their specific preferences and economic interests will be analysed below.

8.4.4 Systematic Regulatory Impact Assessment

Although RIA (*konsekvensbeskrivning*) is obligatory to undertaken within inquiries, there is no requirement for a stand-alone RIA according to a standard format in Sweden, such as in the UK (see Statsrådsberedningen 2000). Instead, RIAs are more integrated in the inquiry reports and can look quite different. Due to the lack of uniform format (with description of the problem, alternative options, etc.), it does not serve as a vehicle for promoting the general procedural rationality of the process in the same way.

In this case, RIA was performed in the 2001 inquiry of a weight-based waste tax model at three different rates (100, 400 and 700 SEK per tonne) (Statens Offentliga Utredningar 2002, pp. 145-191) and in the 2003 inquiry of a weight-based waste tax at 200 SEK per tonne and an energy tax model with existing tax rates (Statens Offentliga Utredningar 2005a, pp. 168-175, 257-263) (see a summary of the main results in **Table 42**). The interviewees had mixed views on their quality and usefulness. First, their role was seen as severely compromised given the lack of alternative EPIs and lack of a clear objective to assess against. The results of the RIA could not be compared to a benchmark, which made the overall assessment of a tax as 'suitable' less transparent and more subjective. Second, several interviewees thought that the RIA within the 2003 inquiry was started too late and there was insufficient time to consider all consequences in sufficient detail. Third, many assumptions had to be made to understand the chain of effects that would lead to increased recycling. Unsurprisingly, many stakeholders questioned these assumptions, as well as the selection of a rate of 200 SEK per tonne for the weight-based waste tax model (which had earlier been proposed to be higher). No one was dissatisfied with the level of quantification of impacts, however. For the 2003 inquiry RIA, a consultant had undertaken a comprehensive modelling exercise that produced relatively detailed figures.

Table 42. Summary of RIA results

Impact parameters	2001 Waste Tax Inquiry – Weight-based tax at 400 SEK per tonne*	2003 BRAS Inquiry – Weight-based tax at 200 SEK per tonne	2003 BRAS Inquiry – Incorporating waste in energy and CO2 taxation
Waste management effects			
Total waste generation	Insignificant impact	-	-
Waste sent to material recycling**	+2.9% after 3-5 years***	+2-10% (depending on increased sorting of biological material)	+4% (i.e. 200,000 tonnes)
Waste sent to biological treatment**	+4.8% after 4-5 years***	Small increase	Very small increase
Waste sent to landfill	+6.2%***	-	-
Expansion of incineration capacity	-1.4% in 2003-2009	-	-
Production of electricity from waste incineration (as opposed to heating only)	Marginal effect	No stimulative effect	Significant stimulative effect
Total waste treatment capacity	+6% in 3-5 years***	-	-
Waste imports	Imports from Holland, Denmark and Norway become non-profitable	-	-
Environmental impacts			
CO2 emissions from waste system (and total national emissions)	-2.86% (-0.09%)	-	-65,000 tonnes per year**** (-0.09%)
NOx emissions from waste system (and total national emissions)	-0,33% (0.00%)	-	-
VOC emissions from waste system (and total national emissions)	-9.98% (0.03%)	-	-
Economic impacts			
Tax cost for industrial waste	-	285 million SEK per year	374 million SEK per year
Tax cost for household waste	-	379 million SEK per year	479 million SEK per year
Household waste charges	+120 SEK per year for houses	+12%	+3-27% (+30-400 SEK per year)
Household cost for source separation	-	20 SEK per year (278 SEK if time cost included)	20 SEK per year (278 SEK if time cost included)
State tax revenue	-	-	+660 million SEK (durable effect)
Administrative impacts			
Administrative costs	-	1.65 million SEK per year	1.98 million SEK per year
EU state support approval	-	More complications	Fewer complications

*The 2001 Waste Tax Inquiry assessed also impacts of SEK 100 and SEK 700 tax rates, but only the mid-level is included in this summary.
**Note that the figures refer to increased sorting of relevant waste fractions, that in turn enables increased recycling.
***Percentages have here been calculated in relation to total amount of treated waste in 1998 (10,429 thousand tonnes), not in relation to the amount currently treated by that particular method, see Statens Offentliga Utredningar (2002, p. 320).
****In addition, CO2 emissions would be further reduced through the incentives to expand CPH capacity among existing and planned incinerators, as opposed to heating capacity only.
Sources: Statens Offentliga Utredningar (2002, p. 55, 160-188, 320-322), Statens Offentliga Utredningar (2005a, pp. 172-186, 257-264, 333-349), Ministry of Finance (2005b).

Comparing the results in **Table 42** above, can the choice of the *energy tax model* over the *weight-based waste tax model* in 2005 be explained? Since the 2003 inquiry decided to recommend the former, impacts from that model were more comprehensively assessed. However, the results suggest that the impact on increased recycling was equal or even favouring the waste tax model, especially if the possible impacts on waste generation would be considered too[90]. In terms of environmental effects and CO2 emissions especially, no assessment was made of the waste tax model. The lack of valuation of the environmental benefits of recycling and CO2 emissions meant that a form of Cost-Benefit Analysis (CBA) was not undertaken. This is not surprising given the disagreement regarding the actual environmental benefits of increasing material recycling (which some meant could for certain waste streams be costs rather than benefits). Neither was a form of Cost-Effectiveness Analysis (CEA) explicitly presented, since the costs on industry and households of the two models were not compared in relation to units of recycling increase. Again, this is not surprising given that the recycling objective was expressed in such vague terms and was not quantified. The cost estimates reveal, however, that the waste tax model would have been more cost-effective as long as the increased material recycling effect would have been 3% or higher (the range had been estimated to be 2-10%)[91]. Regarding administrative impacts, it was concluded that the energy tax model was more likely to get approval from a state support perspective by the European Commission.

Instead of cost-effectiveness or environmental effectiveness, a practical-legal criterion was in the end decisive for the 2003 inquiry committee to propose the energy tax model over the weight-based model. The main argument for the choice between the two was that

> “it is appropriate to deal *first* with the distortion that currently exists within the areas of energy/waste” (Statens Offentliga Utredningar 2005a, p. 41).

The choice was thus not based on the comparative merits of the two instruments, but that it was more ‘natural’ to harmonise the energy taxation by incorporating waste as a fuel and then possibly introduce a ‘new’ weight-based waste tax on top.

Overall, the documents and interviews suggest there were no perceptions of major flaws in the quality of the RIAs that were performed, especially considering the complexity of the waste management market and how many actors would be involved in transferring economic incentives further. The problem here was not as high-cost as for example the landfill targets were to the UK, thus motivating a detailed RIA and CEA. However,

[90] A weight-based waste tax would have provided a stronger incentive to reduce overall waste generation, but the potential effect was considered negligible by the inquiry secretariat and many stakeholders and was not examined in detail.
[91] This conclusion is based upon calculating the total industry and household tax cost per 1% increase in recycling, as quoted in the table.

their role in promoting consideration of wider range of EPIs was prevented by the fact that a tax instrument had been the only realistic option from the start of the process.

8.4.5 Wide consultation

In the incineration tax process, there were three modes for communicating stakeholder interests and ideas; through participation as experts in the inquiries, through written consultation responses, and through lobbying. Regarding the first mode, both the 2001 and 2003 inquiries involved representatives for manufacturing industry, the waste management industry (incineration and recycling industries), and relevant central government authorities (see Statens Offentliga Utredningar 2002 p. 3; Statens Offentliga Utredningar 2005a pp. 3-4). The latter inquiry also included a representative for the association of local government and experts from academia. Most of these interviewed experts were satisfied with the *process* of deliberation in the 15 inquiry committee meetings, but some were disappointed in the lack of consensus-building:

> "I think the climate of discussion was good and everyone got to say what they wanted to say on the tax and the underlying problem… But we didn't really get anywhere. Everyone tended to re-state their basic understanding of the problem and not contribute to more constructive problem-solving"
> "Sometimes it seemed like the chair of the inquiry was deliberately passive in order to let everyone get their concerns 'off the chest'. Of course, he had his hands pretty much tied by the directive to design a tax, no matter what we thought of the need for such a tax… Honestly, it is not one of the better inquiries I have participated in"

As a one-man committee, the chairperson was in the end responsible for the conclusion and proposal. In the final proposal to introduce a tax, the experts were not influential, in that most of them submitted 'dissenting opinions' to the final report. These included both those who opposed a tax and promoted other EPIs (manufacturing industry, mainstream waste management industry, local government) and those who advocated the waste tax model or a tougher tax (recycling industry, SEPA) (see Statens Offentliga Utredningar 2005a, pp. 359-380).

The consultation rounds following the two inquiry reports elicited a high number of written responses, 75 organisations responded to the 2001 inquiry report (Ministry of Finance 2002) and 72 to the 2003 inquiry report (Ministry of Finance 2005a)[92]. The

[92] 82 organisations were invited to respond to the 2001 inquiry report, and 12 spontaneous responses were submitted. 88 organisations were invited to respond to the 2003 inquiry report, and 11 spontaneous responses were submitted.

respondents included central government authorities, some local authorities and county boards, trade associations, individual firms, judicial instances, academia, and some NGOs. As for their influence, the majority of the respondents disputed or were dissatisfied with the problem definition and stated policy objectives underlying the proposal, and several were frustrated by the lack of examination of alternative EPIs. Evidently, these comments were not particularly influential since after the 2002 consultation round a new inquiry was started and after the 2005 consultation round a bill introducing the tax was prepared.

More importantly, a majority of the respondents to both inquiries were directly negative or hesitant to the proposal to introduce a tax, as shown by the response frequencies compiled in **Table 42**. The review of the consultation responses also showed that several of those organisations that were positive to a tax in the second consultation preferred a weight-based waste tax model over the energy tax model, or a combination of both (Ministry of Finance 2005a). Even though all consultation outcomes are not binding and all responses do not carry equal political weight, it is still significant that the government decided against the recommendations of the majority of consultees. Evidently, the party-political pressure to adopt a tax from the Green and Left parties was stronger, and this will be explored further below.

Table 43. Consultation outcomes

Type of respondent	Supported tax proposal	Mixed response*	Rejected tax proposal	No comment
Consultation on 2001 Waste Tax Inquiry report				
Central government authorities	2	6	2	6
Judicial bodies				
Municipalities	1	5	8 (including SALA)	1
County boards		1	1	
Trade associations	6	3	9	2
Individual firms	2	2	9	
Academia	3	1		2
NGOs		1	1	1
Total	**14**	**19**	**30**	**12**
Consultation on 2003 BRAS Inquiry report				
Central government authorities	3	5	3	7
Judicial bodies	1	1		
Municipalities	3	2	10 (including SALA)	
County boards		1	1	1
Trade associations	4		6	2
Individual firms	3	1	11	
Academia	3	1		2
NGOs	1			
Total	**18**	**11**	**31**	**12**

*Respondents were hesitant, noted advantages and disadvantages, noted conditions that should be considered, and/or called for further investigation and RIA.
Sources: Ministry of Finance (2002); Ministry of Finance (2005a).

In summary, there were no complaints about the *process* of consultation and the opportunities to propose and have opinions about EPI choice. However, it is clear that the consultations were not influential, insofar as a tax proposal was put forward despite significant opposition.

8.4.6 Use of evidence

The evidence base appears to be rather substantive in this instrument choice process, even though there were some complaints from interviewees on insufficient knowledge about the effects. One of the key purposes of the system of inquiries is to collect evidence and provide policy analysis. The 2001 and 2003 inquiry reports were indeed comprehensive in scope (both around 500 pages) and built upon commissioned studies of various aspects of the problem and proposal. In contrast to the LATS case, there was also some evidence of an *ex post* nature available, since incineration taxes were used in several European countries: parts of Belgium, Denmark, Italy, the Netherlands, Norway and Austria (Regeringens proposition 2006, p. 33). In particular, the Danish tax, introduced in 1987, was studied in more detail to inform the Swedish proposals.

In line with the fundamental disagreement over problem definition and policy objectives, however, conflicting evidence was referred to by stakeholders on several issues. For example, predictions of the future need for waste management capacity were questioned (see Naturvårdsverket 2000a; Helker Lundström and Axelsson 2005; RVF 2005a), as well as the technical capacity to recycle more plastics (see Statens Offentliga Utredningar 2005a, p. 180). Conflicting LCA studies of the comparative environmental performance of incineration vs. recycling of different waste streams were also referred to (see e.g. Finnveden, Björklund et al. 2005, Carlsson in Statens Offentliga Utredningar 2005a ch. 12, RVF 2003a). The difficulties involved in drawing conclusions from these studies were aggravated in that the studies had different system boundaries and used different alternatives. For example, the results depended on if comparisons were made with landfill or not, if fossil fuels or biofuels would replace waste, and if recyclates would replace virgin raw material.

The interviews confirmed that the problem was not primarily a lack of comprehensive and valid evidence, or necessarily that it was conflicting, but the lack of a clear policy objective and identification of alternative EPIs.

> "You can investigate and investigate for years… In the end, it is always a question of political will. And in this case it was very clearly to introduce a tax – so the analyses were always going to be less important to them".

8.4.7 Summary assessment

It is clear that the level of procedural rationality that was sought and achieved in the incineration tax policy process was severely limited. In contrast to the LATS process, there was persistent disagreement over the problem definition. The objectives were unclear and partly conflicting, and the specified criteria did not have a central place in the process. This meant that there was no commonly agreed basis for identifying alternative EPIs, and thus the tax proposal was not comprehensively and systematically compared with other options. This meant that the RIA was useful only for assessing the 'suitability' of one option, rather than informing the choice between alternatives. The early *a priori* commitment to a tax instrument also meant that role of consultation and evidence gathering for choosing between alternative EPIs was limited. The assessment of the elements of procedural rationality is summarised in **Table 44**.

Table 44. Overall assessment

Element of procedural rationality	Evidence in the incineration tax process
Linear sequence of decision-making stages	The proposed instrument was discussed from the earliest stage, with the problem definition shifting along the way. There was a lack of agreement among key stakeholders over the problem definition and policy objectives, which meant that these stages were not concluded before alternative instruments were considered. There were no distinct or sequential stages.
Clear policy objective and choice criteria	A wide range of objectives, several of which were unclear, ambiguous, and qualitative, were included during the course of the process, i.e. not preceding identification of the alternatives. There was no explicit prioritisation of the objectives, which also involved goal conflicts.
Several alternative instruments identified	The a priori commitment to a tax instrument meant that few alternatives instrument types were considered, and in the end they were seen as 'complementary' rather than alternative.
Regulatory Impact Assessment	Comprehensive RIAs were only performed of the different tax designs considered.
Consultation	Wide consultation, both in written format and through participation in the inquiry commissions. The outcome was not consistent with the final decision.
Use of evidence	The evidence base was comprehensive, although partially conflicting. Ex post evidence collected from other European countries experiences with incineration taxes.

The overall impression of several of the interviewees was that the analysis performed was to a large extent a *post-hoc* rationalisation of an *a priori* commitment to introduce a tax. Unlike the LATS case, it is more difficult to deduce the instrument choice finally made from the stated policy objectives. It was shown above that increased material recycling, to an unspecified level and of unspecified waste streams, could be addressed by a variety of instruments. If the main objective was to contribute to the green tax shift, on the other hand, a procedurally rational process would have implied a comparative assessment of different environmental tax bases, not just incineration. In the end, however, a key objective was stated as harmonising energy taxation and then the chosen energy tax model became the only possible option. Majone’s observation quoted earlier (see section 3.3) thus seems illustrative for this case; “the choice of means helps to alter the criteria by which the correctness of the means must be judged” and “policy goals are often defined in terms of the available means” (Majone 1989, p. 115).

What might explain the low degree of procedural rationality? Like in the LATS case it is possible to identify a set of circumstances relating to the problem setting, beside the political factors that will be explored in the remainder of this chapter. First, it was argued that the Swedish RIA does not function as a vehicle for overall procedural rationality in the same way it can do in the UK. Second, the incineration tax was a proactive national initiative that involved performing ‘better’ than required by EU Directives, meaning that a problem definition and policy objectives had to be agreed upon internally. Third, the costs implied by the intended tax were not of the same magnitude as the costs implied by the landfill targets, which may have decreased the need to demonstrate rational problem-solving to stakeholders affected. Fourth, since incineration also plays a key part in the Swedish energy system, it meant that two complex policy areas had be considered jointly to a greater extent than in the England case. Finally, since stakeholders were involved in the debate on the incineration tax for a long time period and in a more intense fashion in the inquiry committee meetings, there may have been less of a need to demonstrate procedural rationality as a means to secure acceptance of the policy output.

Importantly, though, while the procedural rationality was low, a so-called NEPI was nevertheless adopted. Thus, more procedurally rational processes may not be a necessary condition for the Swedish government to deliver NEPIs and a more diverse

EPI mix. The rest of this chapter will analyse why and how a process that cannot be described as ‘depoliticised’ still resulted in a NEPI. In order to understand the broader political and institutional context of the formal EPI choice process reviewed above, the macro-, meso-, and micro-level framework will be applied (see **Table 16** in chapter 4). How can the political and institutional factors derived from the instrument choice literature help explain both the choice outcome, i.e. a tax instrument, and the low degree of procedural rationality?

8.5 Macro-level factors influencing the tax process

8.5.1 Legal framework

As described above, the incineration tax was a Swedish domestic initiative and not a response to EU legislation. At the national level, no legal or constitutional constraints on the choice of EPI type were referred to in the process. However, in the design of the tax and its exemption, the *EU state support rules* were an important factor. These rules stipulate that all domestic taxes that involve exemptions or rebates (i.e. indirect state subsidies) for certain industry sectors and/or other actors need to be approved by the European Commission to safeguard fair competition on the internal market (Statens Offentliga Utredningar 2005a, pp. 83-90). It was clear from an early stage that a new waste incineration tax would have to be subject to such approval from the Commission, just like the existing energy tax system and landfill tax, since some wastes would have to be exempted from taxation. The 2003 inquiry committee predicted that it would be easier to secure approval of the energy tax model than the weight-based waste tax model. The former could just be notified to the Commission as an amendment of the existing energy tax system, which had already received approval. The latter would require a new application to the Commission and a lengthy process could ensue (*ibid*. p. 182). On this basis the EU state support rules were a decisive factor in the 2003 inquiry’s recommendation of the energy tax model, which the government chose to follow in the 2006 bill (Regeringens proposition 2006, p. 37). Some interviewees thought that the issue of EU state support approval was given undue weight in the final proposal:

> “It started out as a broad and simple tax to provide a negative incentive for incineration, and waste generation more generally… Then it turned into this issue of how can we best ensure legal feasibility of the tax, and the original purpose and objectives of the tax came in second place”.

8.5.2 Policy style and political culture

In chapter 3, it was found that the concept of national *policy style* has been linked to EPI choice in a number of different ways. Assuming that the typical repertoire of EPIs *defines* the policy style (see Rittberger and Richardson 2003), the choice of a tax instrument in this case can be seen as strongly coherent with the Swedish policy style. It was found in chapter 6 (section 6.2.3) that Sweden was an early pioneer with environmental taxes and charges and currently has more than 20 in use. The lack of an ideological or philosophical justification in the latter part of the process suggests that environmental taxes were by then firmly institutionalised and accepted as a Swedish 'way of doing things'. As explained by one of the interviewees,

> "I guess we are just used to environmental taxes in our country, more than in other countries perhaps. I would think that most people by now understand the logic behind them"

Indeed, it was argued above that taxes are not really a 'new' EPI (i.e. a NEPI) in Sweden (cf. Jordan, Wurzel et al. 2003b, p. 16). Possibly, tax instruments are now gradually becoming 'traditional' EPIs, if not yet default EPI choices.

Different dimensions of national policy style can also *explain* EPI choice, according to the instrument choice literature. The Swedish policy style has been characterised as cooperative, consensual, anticipatory, managerial and rationalistic (see **Table 9**, section 3.6.2). Peters (2000) argues that a *managerial* public management style is conducive to choosing complex and discretionary instruments, while a legalistic style is conducive to choosing command-and-control regulation. Arguably, a tax on incineration is more discretionary and less prescriptive than a detailed regulation on which waste streams should be incinerated and not, in that it allows the target actor to make the decision whether to pay or incinerate. Furthermore, Andersen (1999) argues that a *consultative* style is more likely to result in open framework regulations rather than stringent command-and-control regulation. As described above, the highly consultative Swedish policy style was well reflected in the tax incineration process. For the same reason that a tax could be seen as 'discretionary', it could be seen as an open and flexible rather than stringent form of regulation. However, neither of these two propositions are very helpful for understanding this case.

In the England case of the LATS, it was seen that the typically reactive British policy style was helpful for understanding the initial resistance towards taking any policy action, the early preferences for traditional regulation, and the subsequent importance of a cost-minimising instrument (in that case, a TPS). The Swedish policy style has been characterised as *anticipatory* – does this feature explain the choice of a tax? The decision to intervene in the waste management market to faster move up the waste hierarchy and the decision to go beyond requirements from EU waste legislation, seem consistent with a general anticipatory style. Sweden is often seen as an environmental forerunner (although several other countries had already implemented incineration taxes), and is also keen to protect that image (Lundqvist 2004). Although a couple of interviewees questioned the need to be proactive in a European comparison, the argument that Sweden was already performing well on waste management figured surprisingly little in the debate on an incineration tax. There appeared to be a consensus that Swedish waste management should be in the vanguard, and policy instruments had a role to play:

> "We have developed a good infrastructure for waste in this country, based on sound technologies, but political pressure has also contributed… It is in everyone's interest to have a modern and clean waste sector"

Instead, most of the criticism referred to conceptual flaws or practical concerns. In any case, there was no *unitary* anticipatory or reactive policy style reinforced by the government and a large majority of stakeholders such as in the LATS case. Instead, there were two opposing stakeholder groups that advocated different ideas and perspectives on waste management and how best to steer it (see section 8.6.2 below).

The Swedish policy style has also been characterised as *rationalistic*, in terms of a strong belief in the use of knowledge and secular rationality to shape the environment (Richardson, Gustafsson et al. 1982; Ruin 1982). This appears to be contradictory to the low level of procedural rationality in the incineration tax process. However, there are a couple of plausible explanations. First, this case may be atypical in that party-politics effectively precluded the clarification of policy objective and systematic consideration of alternative EPIs, which a majority of the stakeholders called for and considered to be 'standard practice'. Second, a different kind of rationality than procedural rationality may be referred to here. The strong belief in the use of knowledge was evident in the process, in that a considerable amount of evidence was brought in and processed. The 2003 inquiry meetings were also characterised by exchange of knowledge and discussions in terms of 'rational' problem-solving and 'systems thinking'. There was a considerable pressure on the inquiry committee to demonstrate a well-informed proposal. However, the rationality assessed in the previous sections is of a *procedural* kind, i.e. doing the analysis in a certain order to maximise the potential that the decision is well-informed. Possibly, it is not an important feature of the Swedish policy style to demonstrate procedural rationality in order to gain legitimacy of the decision, due to cultural reasons or because stakeholders are generally involved from an early stage.

Related to the 'macro-level' national policy style, it has been proposed that the *political culture* matters for instrument choice, in that *'statist'* countries accept more intrusive (coercive) instruments and centralised government intervention. Linder and Peters (1989) do not unpack what 'statist' might mean, but Sweden is highly statist if measured in terms of size of the public sector in the economy and the general tax burden on society (see footnote 70, p. 209 in this thesis). The proposed incineration tax was intrusive in that it would take resources from the target actors and not hypothecate them. Although there is some circularity to this argument, it may shed some light on how it was possible for the government to surrender to party-political pressure and go against the recommendations from a majority of the stakeholders. First, some interviewees perceived that by the 2003 inquiry the burden-of-proof had been shifted onto the opponents of the tax rather than *vice versa*;

> "Because of the formulation of the directive, it was more a case of finding possible reasons against a tax rather than proving it was a good idea"

Second, much of the critique from opposing stakeholders was not about the general unlawfulness of a tax that would rob them of financial resources, but whether the costs justified the environmental benefits and whether it was the most effective EPI for the problem. The relatively high acceptance of a tax in Sweden has been explained in terms of levels of trust, in politicians and as generalised social trust (Jagers and Hammar forthcoming). It is beyond the scope of this thesis to probe further into this issue, but it appears that the statist political culture was conducive to the political decision to go ahead with the tax despite opposition.

Political culture is also a central factor in the study of *ecological modernisation*. It has been stated that new EPIs, such as economic instruments and informative measures, are hallmarks of ecological modernisation, and that an ecological modernisation agenda develops more easily and rapidly in a society with a *corporatist* political culture. Sweden is normally seen as a highly corporatist country, with a clear ecological modernisation agenda. However, attributing the adoption of economic instruments such as environmental taxes to corporatism is dubious, given that both industry and trade unions have resisted for example energy taxes and that generous exemptions have been made for the industrial sector which is vital to Sweden's economy. Overall, the lack of a developed and clear theoretical linkage between corporatism, ecological modernisation and economic instruments makes it a difficult explanation to apply on the case of the incineration tax.

To conclude, this analysis shows that adopting an incineration tax was consistent with the statist political culture and some aspects of the Swedish policy style. The statist political culture explains the general acceptance of taxation *per se*. The consultative and managerial policy style could explain the choice of a tax rather than detailed command-and-control regulation. However, the existence of stakeholder groups advocating both reactive and anticipatory approaches to the problem suggest that a macro-level, national policy style did not go unchallenged. Finally, insofar as the EPI repertoire defines the national policy style, the incineration tax was not challenging the Swedish policy style since environmental taxes are increasingly becoming 'traditional' EPIs in Sweden. However, none of these factors appears to have been active or immediate driving forces upon the choice to introduce an incineration tax.

8.5.3 Party politics and ideology

It has already been described that factors related to *party politics* were a key driving force on the adoption of the incineration tax. Here, the role of ideology will first be discussed. First, what was the reason for the *a priori* commitment to a tax instrument that was seemingly made in the early 1990s? The first government inquiry in 1990 had the explicit motive to identify potential environmental tax bases, and the underlying rationale was dual; to apply the polluter-pays principle and increase the economic efficiency of environmental policy, by internalising externalities and achieving least-cost pollution reduction (Statens Offentliga Utredningar 1990). These rationales appear to have been widely accepted as the following inquiries did not motivate an incineration tax so much with reference to these underlying principles (Statens Offentliga Utredningar 1994; Statens Offentliga Utredningar 2002). Furthermore, the rationale of using environmental taxes for raising revenue also became more legitimate, beginning with the 1991 energy tax reform (when income taxes were lowered) and culminating with the 2001 green tax shift (see below).

It is significant that the incineration tax was never explicitly analysed as a market failure and it did not build on an estimation of the externalities of incineration (like the incineration tax considered in the UK did, see HM Treasury 2004b) or a CBA indicating what level of increased recycling should be aimed for. Thus, the economic efficiency rationale of environmental taxes sometimes associated with neoliberal ideology did not figure. The right-wing political parties noted this absence in the parliamentary debate on the tax proposal:

> "The government's proposal to tax incineration of household waste will lead to significant increases of waste collection charges at the same time as the environmental effect will largely be absent. In the underlying inquiry the reduction [in incineration] is quoted as c. 4%, i.e. 96% of the waste will be burdened with a tax without there being an improvement, at a tax cost of 600-900 million SEK per year. The bill is yet another example from the Social Democratic government that under an environmentally friendly flag is only intended to collect tax revenues to the state." (Skatteutskottet 2006 p. 10)[93].

The *a priori* commitment to the tax instrument was thus originally a conscious ideological decision, but over time it seems that environmental taxes became less ideologically charged and seen as a policy instrument much like others.

The EPI literature has found that environmental taxes are indeed more popular with left-wing governments (Daugbjerg 2001a), since they more generally support the idea of 'big government' and redistributive measures (Jordan, Wurzel et al. 2003a). This certainly seems to hold true for Sweden and the incineration tax case. The green tax shift was agreed in 2001 between the Social Democratic government and the Green and Left parties and stipulates a shift from income tax to environmental taxes (mainly energy and CO2 tax) worth 30 billion SEK in the period 2001-2010 (see section 6.2.3). According to an interviewee, the need to find further tax bases to progress with the green tax shift was since the early 2000s the most important objective with the tax:

> "The Left Party was pushing for [a new incineration tax within the context of the green tax shift]. The Green Party was a little more cautious, and wanted it to be investigated more".

Virtually all interviewees recognised the green tax shift as one of the most important or *the* most important (unofficial) objective, but were critical or unappreciative as to why it needed to target the waste sector.

[93] This is an excerpt from a Reservation signed by the Moderate Party, Centre Party and Christian Democrats to the Tax Committee's report on the 2006 incineration tax bill.

> "I can understand the green tax shift as a valid political ambition. But that is a political issue… The problem was that here we were trying to *solve a waste problem* primarily, and introducing a purely fiscal tax was not going to help us change behaviour in a more environmentally friendly way."

The formulation of the 2003 inquiry directive shows the reluctance of the government to state the green tax shift as a legitimate purpose for a new tax on incineration, as it was only briefly mentioned. Obviously, an implication of explicitly stating the green tax shift as the primary objective would be that critics would call for comparisons with other potential environmental tax bases, in line with a procedurally rational EPI choice process. In the end, the elevation of the objective to harmonise energy tax legislation and eliminate inconsistencies was arguably a convenient way for the Social Democratic government to draw attention away from the more controversial green tax shift motive.

The EPI choice literature has also found that the presence of Green Parties can help 'tip the balance' in favour of NEPIs (Jordan, Wurzel et al. 2003a). In this case, the Green Party together with the Left Party were instrumental to the adoption of the tax, since the Social Democratic government depended upon them for parliamentary majority from 1998 to 2006. Shortly before the establishment of the 2001 inquiry, the Green Party spokesperson Birger Schlaug threatened that:

> "The Social Democrats have to give in when it comes their view on waste incineration… I am convinced that we can agree with enough parties so that the Social Democrats would have to rely on the Moderate Party [the largest right-wing party] to block it. And that would ruin their strategy when it comes to portraying their ambition to build a 'green welfare state'…" (Miljöpartiet de Gröna 2001)

According to interviewees, the political negotiations between the Green and Left Parties and the Social Democratic government led to the unclear 2003 inquiry directives. The former insisted that a detailed and legally workable tax design should be proposed, while the Social Democrats added that 'other economic instruments', preferably of a 'positive character' (see section 8.4.3), should also be looked into. After the consultation round following the 2003 inquiry, it was the Left and Green Parties that negotiated the announcement of a new tax in the budget bill presented in September 2005. In the context of the larger budget negotiations, the Social Democratic government did not think the incineration tax was a big issue to 'fuss about' (Anon. 2005a).

To conclude, party-political bargaining, made possible by the minority government of the Social Democrats, was crucial to the choice of a tax in this case, and also that it was introduced when it was and not delayed by further investigation. The tax was also a direct result of the green tax shift that the government together with the collaboration parties had agreed upon in 2001.

8.6 Meso-level factors influencing the tax process

8.6.1 Institutional arrangements and organisational culture

The role of institutional arrangements and organisational culture is more evident in the *design* of the incineration tax, i.e. the choice between the energy tax model and weight-based waste tax model, than in the choice of a tax as an EPI *type*. According to institutionalist theories, there is *path dependency* in instrument choice acting as a barrier towards NEPIs, for reasons such as misfit with the system of rules in a policy area, considerable investment of time and resources in 'old' EPIs, professional backgrounds among policy-makers, and habit and routine. The incineration tax as a NEPI appears to have broken such a path dependency. However, it was argued above (section 8.5.2) that the Swedish policy style has developed so that environmental taxes are not really 'new' EPIs anymore and are becoming mainstream instruments. This means that there was an institutional framework for taxes in place. Instead, this argument can be turned on its head; the early interest (in the early 1990s) in a tax to limit incineration and stimulate recycling created a path dependency that excluded the consideration of alternative EPIs, including 'old' EPIs.

A more directly observable influence of institutional arrangements and organisational culture is the promotion of the energy tax model rather than the weight-based waste tax model by the Ministry of Finance (MoF). As mentioned earlier, the MoF was responsible for conducting this policy process since it was a tax proposal. According to an interviewee, the *institutional home* of the issue played a role:

> "It is obvious that the Ministry of Environment had to battle with the MoF to get the recycling and waste objectives into the [2003] inquiry directives… The MoF was more concerned with getting a legally workable proposal that did not conflict too much with other legislation and policy"

When the idea was floated to incorporate waste as a fuel in the existing energy tax instead of preparing new tax legislation[94], some interviewees speculated that the MoF

[94] Note that a weight-based waste tax on incineration would have been incorporated into the existing law on the landfill tax. However, this solution would have involved more work on definitions of waste and definition of exemptions.

lobbied for this idea despite that they should let the inquiry conduct its analysis independently. The MoF and other proponents of this model viewed it as a 'legally elegant' solution that would achieve several objectives; removal of inconsistencies in existing energy tax legislation, approval from an EU state support perspective, provision of incentives to sort out fossil fractions (i.e. plastics) from waste sent to incineration, and provision of incentives to invest in CHP capacity. Skeptics, on the other hand, saw this model as essentially simplifying MoF's work:

> "I never understood why the weight-based waste tax model was dropped towards the end of the [2003] inquiry… Yes, it would have involved more work on defining exemptions and so on, but it is the purpose of the tax that matters and should guide the decision… The idea to combine the energy tax model and weight-based waste tax model was also not investigated properly… There was too much focus on the legal aspects and too little on the environmental and economic aspects, in my opinion"

Professional background may also have played a role. The 2003 inquiry chairperson, the majority of the inquiry secretariat staff and the MoF officials responsible for taking forward the inquiry proposal were all from the legal profession. Overall, the institutional arrangements around environmental taxation in Sweden seems to have influenced the choice of the energy tax model, and also facilitated the adoption of a new tax *per se*.

8.6.2 Policy ideas, learning and transfer

Regarding theories on instrument choice focusing on the role of ideas, it has been argued that instrument innovation is more likely when principles and policy objectives governing a policy field are reconsidered, i.e. when there is *paradigmatic change* (Howlett and Ramesh 1993). In comparison with the LATS case, there was a 'normal period' of waste policy-making in Sweden in the early 2000s and, consequently, there was more of 'fine-tuning' existing tax instruments than innovation.

The role of *international transfer of EPI ideas* has also been investigated in the instrument choice literature. As mentioned above, evidence from the use of incineration taxes in other European countries was collected by the inquiries. While they may have informed the thinking of the investigators, the interviews and policy documents do not suggest that these experiences had a particularly important role or that the Swedish proposal emulated an already tested model.

Ideas did play an important role in the incineration tax process, however, in that two opposing *perspectives on waste policy* among the key stakeholders could be discerned in the case study material, with different implications for instrument choice. These ideas related to waste policy in general, and are therefore best considered as a meso-level factor in EPI choice rather than a micro-level factor. The actors and their ideas came together in the *reactive/pragmatic* perspective and the *proactive/principle-based* perspective, which are outlined in **Table 45** below. The former included the mainstream waste management sector (SWM), local authorities (SALA) and manufacturing industry (CSE). The latter included the recycling industry (SRIA), an environmental NGO (SSNC), and the Swedish EPA. Their ideas were connected to the economic and political interests of the respective actors (which will be explored below), but the ideas in themselves are also important for understanding the disagreements in the process. The ideational differences between the two stakeholder groups prevented effective communication and agreement on the problem definition justifying a tax (cf. Saarikoski 2006). While there was some level of organised interaction and collaboration within the two perspectives (e.g. references to each other's evidence, joint statements to the inquiry, and joint debate articles published in newspapers), they should not be seen as 'advocacy coalitions', as defined by Sabatier (1998, in Jordan, Wurzel et al. 2003c). A more appropriate concept then is Hajer's (1995) 'discourse coalitions'.

Table 45. Competing perspectives on waste incineration

REACTIVE/PRAGMATIC PERSPECTIVE	PROACTIVE/PRINCIPLE-BASED PERSPECTIVE
Waste policy principles	
• The waste management hierarchy is too crude as a guideline, it is more important to consider detoxification and waste properties when deciding between options • Incineration and landfill tax signals are too indirect to influence waste generation • Import of waste is rational when Swedish waste management is more environmentally friendly	• The waste management hierarchy is a valid and good guideline for policy • Waste generation can be influenced through a taxes on landfill and incineration • Import of waste should not take place, but the self-sufficiency principle should apply
Waste – waste or resource?	
• Waste is simply waste and should not be compared to other fuels and energy sources • Incineration is a waste treatment method with a positive by-product: energy • Since the marginal fuel is fossil fuels, waste incineration means that the environmental impact of the energy system is lower	• When used for energy recovery, waste is a resource and comparable to other fuels • Incineration is a waste treatment and an energy production method • Since the marginal fuel is biomass, waste incineration means that more environmentally friendly energy generation is hindered
Principles for public policy intervention	
• The *market* (given existing policy instruments) organises waste management efficiently • Policy instruments should be used to ensure desired changes in the *short term* • When policy instruments are used, they should *target the problem as precisely as possible* • Environmental taxes penalise investment decisions made in good faith and they are a pure penalty when there is no possibility to change behaviour	• There is inertia in the waste management system that justifies *public intervention* • Policy instruments should focus on changes in the *long term* • Policy instruments can be *blunt*, setting out a direction for change and leaving it to the market to make specific decisions • Environmental taxes implement the polluter-pays principle and stimulate and increase the competitiveness of new technologies

First, the two perspectives diverged on general waste policy principles. As has been described earlier, there were disagreements on the validity of the *waste management hierarchy*. There were technically detailed discussions on the comparative environmental performance of incineration vs. material recycling and biological treatment of particular waste streams, but also deeper divisions on the hierarchy's general usefulness. According to the reactive/pragmatic coalition, it was too simplistic and crude to use as a policy guideline. Instead, assessments of individual waste streams were promoted, to determine the best treatment option for each one. Furthermore, it was argued that it led to too much focus on the quantity of waste to be diverted up the hierarchy. As argued by one interviewee,

> "What we mustn't forget is to tackle the *quality* of waste. With purer fractions, recycling is more efficient. You will have a perverse effect if a tax shifts a lot of waste from incineration to recycling, and it is no longer good enough for recycling"

The proactive/principle-based coalition, on the other hand, tended to argue that the simplicity and intuitiveness of the hierarchy was also its strength. They referred to the *inherent* values of recycling over incineration that had also been recognised in the government waste policy (Regeringens proposition 2003, pp. 21-22), namely no landfill of by-products (like incineration ash), energy savings from less extraction of virgin raw materials, the closer linkage to product design, and public awareness of the need to re-use products. Other important differences between the two perspectives related to the consideration of *waste generation* as a valid objective and the *import of waste* for incineration. Within the reactive/pragmatic perspective, waste generation was seen as a separate problem from incineration. Within the proactive/principle-based perspective, on the other hand, even weak negative economic incentives for waste generation arising from an incineration tax was seen as important, not least in terms of the symbolic value. Regarding import of waste, the proactive/principle-based perspective emphasised the self-sufficiency principle that should guide national waste management and the risks involved in developing an import dependency. The reactive/pragmatic perspective, on the other hand, argued that it does not matter where waste is generated, but the key issue is that it can be environmentally better managed in Sweden – and generate electricity and heating. In other words, existing infrastructure should be used optimally. Overall, the reactive/pragmatic perspective wished to see a more precise and specific problem definition, which would likely have resulted in more precise instruments than a tax.

Second, the perspectives and associated stakeholder groups differed in their view on *waste as a resource or as a fuel*. Waste as a resource means that it is an energy-generating fuel comparable to virgin fuels, such as fossil fuels or biomass. The implication is that it is logical to subject waste to the same energy and CO2 taxation as other fuels, in order to steer the national energy mix towards lower-carbon fuels. On the other side was the reactive/pragmatic perspective which viewed waste as waste, with its energy-generating potential as a positive by-product. It was argued that waste is not comparable to other fuels, because it cannot be avoided or replaced. The relevant comparison, according to them, is the environmental performance of alternative waste treatment techniques. Furthermore, they argued that waste had already been taxed at the product stage, hence it would be unfair to tax it again as a fuel.

A related discussion was what the *marginal fuel* in the Swedish energy system was and would be in the future. The proactive/principle-based perspective argued that it was biofuels, hence waste incineration was hindering a more renewable energy source. The waste-as-waste proponents stated that virgin fossil fuels was the marginal fuel, which made incineration preferable. The implication is that a tax on waste incineration would be counterproductive in relation to the climate policy objectives.

Finally, underlying these ideas on waste and energy policy, some differences could be discerned in ideas on *general public policy intervention* and how and when it was justifiable. On a general level, the reactive/pragmatic perspective tended to argue that market mechanisms should determine the waste management mix, and policy instruments should be used to correct specific problems. Their opponents argued instead that the waste management market was not operating efficiently due to imperfect competition, and therefore public intervention was necessary. Furthermore, the reactive/pragmatic perspective employed a shorter-term perspective and wished to focus on particular waste streams that could and should be shifted from incineration to material recycling. Essentially, they were interested in what was feasible, practicable, and environmentally beneficial today, and implied that the government could specify precise and targeted interventions to address these limited problems. Environmental taxes were seen as too blunt and also directly counterproductive when there was no opportunity to change behaviour in order to avoid the tax. On the other side, the proactive/principle-based perspective to some extent had a longer-term perspective and argued that an important purpose of economic instruments is not to steer precise and planned changes, but to give incentives for dynamic effects and innovation. They also referred to the polluter-pays principle as a viable justification for a tax, regardless of its potential effectiveness. They were more accepting towards a blunt and less predictable instrument.

These differences in underlying policy principles meant that it was difficult for the actors involved to communicate and find potential common ground. In the end, the government did not have to choose side or try to establish a consensus. While it did determine that waste was a fuel comparable to others, positioning towards waste policy principles and principles relating to environmental taxation could effectively be avoided through the focus on the harmonisation of energy taxation objective. Instead of making the incineration tax into an ideational decision, it was made more or less into a practicality, although with substantive effects in terms of revenue.

8.6.3 Policy network characteristics

Bressers and O'Toole's (1998) theory on *policy networks* and instrument choice proposes that the more integrated a network is, the less costly and coercive instruments will result. Integration is measured by two network features; *cohesion* and *interconnectedness*. This theory is complex and a full application goes beyond the scope of this case study. However, it can be observed that the cohesion ("the extent to which individuals, groups, and organisations emphatise with each other's objectives insofar as these are relevant to the policy field", p. 219) can be considered weak in this case, given the deep divisions in ideas and policy principles among key members of the waste policy network described above. Interconnectedness, on the other hand, was strong due to the participation of network members as experts in the frequent inquiry meetings and repeated consultations. According to the authors, this would result in instruments with, *inter alia*, 'normative appeal', 'limited withdrawal of resources', and 'absence of freedom for target groups to opt out of the instrument'. While the first and the latter certainly are key features of environmental taxes, it is clear that the strong interconnectedness in this case did not limit the withdrawal of resources. In other words, this aspect of integration in the policy network did not lead to a less coercive or costly instrument for the target actors.

8.7 Micro-level factors influencing the tax process

8.7.1 Actor resources and interests: introduction

The EPI choice literature concerned with explaining a particular instrument choice situation and how the power relationships between government and producers affect the outcome has made two important predictions. Macdonald (2001) has found that the more powerful and organised the producer group, the *less coercive* instrument is chosen. In a similar vein, Daugbjerg (1999) has proposed that producers prefer instruments with the lowest net cost (or highest net benefit), which means that negative economic instruments like taxes are those least preferred. The power of the producers towards the government determines their success in realising their preferences. Clearly, a tax instrument is both a negative economic instrument and coercive, in that a financial 'penalty' follows if behaviour is not changed. Does this mean that the producer group here (i.e. incinerator operators) was not powerful? The waste management industry operating incinerators and their local authority owners and/or clients were organised and lobbied against the tax. Evidently, though, they were less powerful than the Left and Green Parties were in relation to the government. The manufacturing industry, on the other hand, managed to secure an exemption of industrial waste to the tax at a late stage in the process (see further discussion below), and thus seems to have been more powerful.

Arguably, though, the appearance of coerciveness towards the producers could be toned down by the government by focusing on harmonisation of energy taxation as the immediate rationale and purpose of the tax. In fact, the introduction of the tax could be framed as a removal of an unfair and environmentally harmful 'indirect subsidy' or 'tax exemption' of waste as a fuel. The tax bill motivated the choice by stating that:

> "In general, consistent taxation means that there is a lower risk of competition bias. From a climate perspective, the tax exemption implies that CO2 emissions are valued differentially for different fossil fuels, even if the emissions occur within the same sector. This tax exemption may mean that waste incineration is larger than what is economically efficient." (Regeringens proposition 2006, p. 36).

Although the official primary objective was still stated as increased recycling in the bill, the framing of the tax as eliminating an existing tax exemption made it appear less coercive.

Like with the English LATS case, the EPI choice process in this case can also be analysed as a process of bargaining between actors to protect economic and political interests. Below, economic and political interests are analysed in relation to three different aspects of instrument choice; the problem definition, the choice of EPI type, and the decisions on design features of the instrument. As concluded above, these were not sequential stages in the process, but the division is still analytically helpful.

8.7.2 Political and economic interests in the problem definition

As described above, the persistent disagreement on problem definition in this case led to unclear objectives of a tax and, in the end, a tax bill in which the government effectively avoided to define the problem more precisely (except for defining waste as a fuel). Part of the disagreement among stakeholders originated from fundamental differences in ideas and perspectives on waste policy, but the respective ideas of the opposing groups were also linked to their economic interests. The mainstream waste management industry would bear much of the direct tax burden, and transfer it onto households by increasing the general waste collection charges. Waste incineration was also a common practice within manufacturing industry, so they too would also be losers of a tax. The winners in this case were recycling firms, for whom new business opportunities would result from more competitive prices.

To minimise the losses, several arguments against a tax and its underlying rationale based on issues of equity and competitiveness were raised. First, the mainstream waste management and manufacturing industry were concerned over the purely 'penal' effect of a tax, when there was a lack of recycling capacity to divert waste from incineration to. One interviewee argued that

> "The tax is just not the right way to achieve increased recycling here… The point with an environmental tax, as I understand, is that it should give an incentive to change something. But it is impossible to change much of the current waste management, and then you are faced with just having to pay a penalty despite you are doing the best you can"

The recycling industry, on the other hand, argued that the market did not work as efficiently as the mainstream waste management industry suggested, due to imperfect competition and a lack of knowledge among local authorities about possible recycling options. SRIA stated that

> "Are the local authorities aware that the plastics recycling firms face a shortage of supply of waste and have to import plastics waste to be able to deliver recycled raw material to industry?" Helker Lundström and Axelsson 2005

Furthermore, they pointed out the risk of technological lock-in with increased incineration, which would limit their opportunities to grow as an industry.

> "The expansion of incineration capacity means that we are locking ourselves into an infrastructure that blocks the development of a sustainable society" (Helker Lundström and Axelsson 2005)

In addition to pointing out the technical and market conditions that would prevent the intended tax from working efficiently, local authorities and the manufacturing industry also argued that the tax would lead to inequitable and unacceptable costs that would be harmful to the economy and politically risky. The manufacturing industry threatened it could harm Sweden's competitiveness:

> "A national tax would also means that industry can lose market share since increased costs in these activities [where incineration of process waste is the best environmental option] cannot be transferred onto the price of the products" (CSE in Statens Offentliga Utredningar 2005a p. 370).

The local authority association and the waste management industry together pointed to political problems and increased costs for households:

> "Depending on the waste management system in each local authority a transition to a new tax principle will involve large differences between the authorities in the increase of waste collection charges that the tax proposal leads to… We think that the investigator ignores the difficulties that can arise when you locally have to justify why waste collection charges have to increase by as much as 25-30%, sometimes even more. This increase will occur despite the local authority having invested in source separation of food waste and taken action to introduce source separation of packaging and paper so that the producer responsibility targets can be achieved" (Statens Offentliga Utredningar 2005a, pp. 359-360).

Finally, the actors who would lose out from the tax also questioned whether the estimated environmental effect really justified the costs:

> "According to the inquiry's own figures the Swedish CO2 emissions would decrease by around 0.1% as a consequence of the tax. According to my own assessment, the very marginal reduction cannot motivate the introduction of a tax considering the additional administration and costs for businesses and government authorities." (CSE in Statens Offentliga Utredningar 2005a p. 370).

Many of these arguments were picked up in the subsequent party-political debates on the tax proposal. However, due to the 'higher-level' political bargaining between the Social Democrat government and the Green and Left parties on the overall government budget, the tax became a minor party-political issue for the government rather than an issue for which they had to engage heavily in interest group politics.

8.7.3 Political and economic interests in the choice of EPI type

The opponents of the tax thus first disputed the problem definition underlying the tax, but also tried to persuade the government to consider alternative EPIs. It was seen in sections 8.4.3 that the stakeholders proposed a range of alternative EPIs in their consultation responses. A review of these consultation responses (see Ministry of Finance 2002; Ministry of Finance 2005a) shows that much of the opposition against a tax stemmed from the view that there was enough government intervention in waste management already, rather than the view that another EPI would do a superior job. The alternative EPIs that were suggested were often thus a 'second-best' solution for the respondents in question.

A *strengthened producer responsibility* (in particular for plastics) was the most commonly mentioned alternative. The support from the waste management industry, local authorities and district heating industry was based on the principle to address the waste problem at its source. In a dissenting opinion to the 2003 inquiry report, these actors argued that

> "[the proposed tax of the fossil content of waste] means a deviation from the principle that producers are responsible for more environmentally friendly products and achievement of stated material recycling goals. This principle has guided waste policy for more than 10 years." (SWM and SALA in Statens Offentliga Utredningar 2005a p. 360).

However, it would of course also place the direct financial burden on producers rather than the waste management industry.

These stakeholders also expressed interest in the principle of *taxing products and raw materials*, to stimulate the demand for recycled material. Also such an instrument would place the immediate cost burden on producers rather than the waste sector. A *charge on waste incineration* rather than a tax was proposed by some local authorities. This would recycle revenues back to the them rather than financing other parts of the government budget. A more short-term measure that was proposed by the mainstream waste management industry and SEPA was *investment grants for biological treatment*. This was seen as more environmentally friendly and with a larger capacity potential than material recycling and composting. *R&D grants* to recycling technologies and *education and information* initiatives were also supported by a broad set of stakeholders.

Some recycling firms, the environmental NGO and academia, on the other hand, supported the incineration tax. However, it was seen as insufficient in providing strong incentives for increased recycling. Several of these actors thus argued that the weight-based waste tax model could be introduced in combination with the energy tax model.

It is thus clear that the first preference of the target group of the intended tax, i.e. waste management companies and manufacturing industry with their own incineration facilities, was no new EPI at all, i.e. a no-action alternative. As a second best option, the waste management industry (as opposed to the manufacturing industry) argued that an EPI should be directed at producers instead. As a third best option, various 'soft' or positive economic instruments were proposed. These preferences correspond with

Daugbjerg's (1999) stipulated order of preferences of producers, and the actors thus seemed to protect their economic interests when proposing alternative EPIs. As argued above, it was the party-political pressure on the government that led it to decide to introduce a tax in 2005 and to go against the recommendations of several stakeholders, as well as not pick up their proposals for alternative EPIs.

8.7.4 Political and economic interests in the design of the instrument

Compared to the LATS case, there was not as much political bargaining after the decision on EPI (including the choice of the energy tax model over the weight-based waste tax model), which in this case can be seen as the period after the 2005 government budget negotiations. Two important issues emerged, however; the *exemption of industrial waste* from the tax and the *measurement methods* to be used for determining fossil content of household waste.

The final inquiry report published in March 2005 had proposed that the tax should apply to both household and industrial waste. The latter had been a tricky issue, in terms of definitions of relevant wastes and incineration as a recycling method, and led to lengthy discussions on various exemptions. As described above, representatives for industry were persistently very critical to the tax. Still, the government announcement of a new tax in the 2005 budget bill did not indicate that industrial waste was to be exempted. Sometime before the March 2006 bill proposing the new tax legislation, however, the decision was made to exclude industrial waste[95]. This exemption meant that a tax cost estimated at 374 million SEK (42 million EUR) per year could be avoided by industry (see **Table 42**). Consequently, it also meant that state tax revenues would decrease. The originally estimated 660 million SEK (73 million EUR) revenue per year, was in the 2006 bill decreased to 410 million SEK (46 million EUR) per year (Regeringens proposition 2006, p. 67). This also meant that progress with the green tax shift – arguably the most important yet unofficial objective of the tax – would be slower. Finally, the effect on increased recycling (previously estimated at 4%) would be lower, but the bill did not provide a new estimate. The Green and Left Parties did not raise any

[95] The interviews for this case study were conducted in September-October 2005 and did not therefore address this issue and its implications.

objections in the parliamentary debate on the tax bill, so it is unclear whether this was a joint decision or a victory for the Social Democrat government.

The problems with specifying measurement methods for the fossil content of household waste meant that the tax was delayed. In the 2005 budget negotiations, the Green and Left Parties had bargained for a quick process to present a bill which would allow the tax to enter into force by 1 January 2006. However, the SWM lobbied intensively for a more detailed consideration of measurement methods. The end result meant that the proxy for determining fossil content of household waste was lowered somewhat, from 14% to 12.6% for household waste (Regeringens proposition 2006, p. 43). The SWM were pleased that they managed to address this issue and in the process of doing so delaying the introduction by 6 months:

> "We view it as a success. There were big holes in the original proposal, which the government was finally obliged to admit… The postponement also saved the members a couple of hundred million" (Wiqvist in RVF 2006a p. 4).

Compared to the LATS process, there was thus less interest group politics involved in the determination of some design features, although the exclusion of industrial waste was a decision with major impacts on the environmental effect of the EPI and its revenue-generating potential. However, overall this case study shows that there were more politics involved in defining the problem and clarifying the objectives of the tax, compared to the LATS case.

8.8 Conclusion

Before drawing conclusions on the key factors shaping the EPI choice process studied in this case, how has the resulting incineration tax worked in practice so far? The tax was introduced on 1 July 2006 and there is no information yet available on its early effects. The bill stated that the tax should be evaluated after its introduction, *inter alia*, to assess whether complementary instruments were necessary. No date for such an evaluation was set, though. Depending on the evaluation outcome, the government has stated that the tax base could be extended to include industrial waste and that the tax may need to be reconsidered depending on the future of the EU Emissions Trading Scheme (Regeringens proposition 2006, p. 44, 56).

In this case, a so-called NEPI was adopted, and more specifically an environmental tax. However, it was seen that this choice did not result from a conscious and procedurally rational search for alternative EPIs to address a given problem. Instead, the process was characterised by persistent disagreement over the problem definition; unclear, conflicting and partly unofficial policy objectives that changed over time; little systematic assessment of alternative instruments; and, consequently, a limited RIA. The use of evidence and provision of consultation opportunities were better, although the consultation outcome was not influential in the end. Overall, most interviewees were very dissatisfied with how the process was conducted and EPIs considered. It was seen as a case of an *a priori* commitment to a tax instrument, with a *post-hoc* rationalisation in terms of the late addition of the harmonisation of energy taxation objective. However, other stakeholders pointed out that this was the first, badly needed policy instrument to increase material recycling.

A set of case-specific circumstances were identified as possible explanations of the low procedural rationality as compared with the England case presented in the previous chapter. These included the lack of a standard format for RIA in Sweden, the domestic origin of the initiative (as opposed to responding to EU legislation), the relatively low cost of the problem, the complexity involved in joint consideration of the waste and energy systems, and the lower need to secure acceptance of the tax due to the involvement of stakeholders in government inquiries.

However, it was found that the general lack of procedural rationality – as well as the choice of a tax instrument – can also be explained by political and institutional factors. At the macro-level, it was found that the general ideology of environmental taxes as instruments was widely recognised (if not accepted by everyone) and that taxes were consistent with the national policy style and political culture. Thus, environmental taxes were not really seen as 'new' EPIs. However, the immediate and decisive reason why the incineration tax was adopted was the strong party-political pressure from the Left and Green Parties on the Social Democratic government to secure new tax bases for the green tax shift. Within this macro-level context, it was seen that stakeholders still engaged in advocacy of ideas and principles for waste policy, in attempts to propose alternative EPIs that would better protect their economic interests, and in political bargaining to mitigate the effects of the tax. Towards the end of the process, the new emphasis on the need to harmonise energy taxation, by including household waste as a fuel, was convenient both institutionally and politically. Institutionally, it meant less work for the Ministry of Finance to seek EU state support approval. Politically, it provided the Social Democrat government with a more legally-based reason to go against the consultation outcomes and to offer a compromise solution to the target actors compared with a more costly and comprehensive weight-based waste tax.

When drawing conclusions from this case study, it should be emphasised that the incineration tax represents a more ambitious and proactive policy intervention than the England LATS instrument, with reference to aspirations to move upwards in the waste hierarchy. Nevertheless, it can be concluded that this was far from a depoliticised EPI choice. There was considerable opposition to the tax, but also those that were disappointed with the compromise outcome. This case suggests that NEPIs can be chosen also in highly politicised policy processes. It also confirms the finding from the previous chapter that much of the politics around EPI choice is played out not necessarily in the (lack of) debates on alternative EPI *types*, but in the debate on underlying problem definition and the detailed design of the chosen instrument type. For example, in this case the selection of the energy tax model over the weight-based waste tax model and exemption of industrial waste at a late stage will have significant effects on the actual impact of the instrument.

Chapter 9

Conclusion: Diversifying EPIs and Beyond

9.1 Introduction

In the introduction to this thesis, it was argued that a trend of depoliticisation of EPI choice can be discerned. Such a trend has been noted in the theory on EPI choice, in that previous ideological biases and preferences favouring certain EPI types (e.g. environmental taxes, command-and-control regulation, voluntary agreements) among different environmental policy stakeholders have gradually eroded (see e.g. Cook 2002; Golub 1998b). Furthermore, it has been proposed that governments increasingly need to ensure legitimacy of policy decisions through demonstrating a technical, professionalised and procedurally rational decision-making process that systematically considers alternative policy instruments (Howlett 2000b). It was also described that this trend is visible in policy practice, in that international organisations and national governments have committed to using a wide variety of EPIs, and especially so-called NEPIs. These commitments also refer to comparative assessments of alternative EPIs in the decision-making process, in order to find the 'best-tool-for-the-job'. Skeptics, however, have called this trend nothing but 'naïve instrumentalism' (Böcher and Töller 2003).

With the broader issue of depoliticisation as a context, two research questions were formulated, one focusing on outcome of EPI choice and the other of process of EPI choice:

- *Has the mix of EPIs for municipal waste policy become more diverse over time?*
- *What determines the instrument choice process? In particular, to what extent does the ideal of procedural rationality influence the consideration of alternative EPIs?*

In this concluding chapter, the empirical findings from EPI choice in the municipal waste policy area in Sweden and the UK (England) will be summarised. The extent to which these findings can be generalised, to other countries and to environmental policy as a whole, will then be discussed, as well as the implications for future EPI choice in the waste policy field. Based on these conclusions, new issues arising from the research that need to be considered in further theoretical understanding of instrument choice are addressed. Finally, this chapter is concluded by outlining a set of specific issues for future research as well as policy recommendations.

9.2 Progress in EPI diversification – but is it slowing?

Chapter 2 described how the use of various NEPIs had increased considerably during the 1990s and early 2000s, although there was disagreement whether this increase had been fast or slow, and whether NEPIs had replaced traditional environmental regulation or complemented it. In line with recent literature on the need to study instrument mixes, the decision was made for this thesis to not examine the use of a particular type of EPI but rather identify all key instruments in a certain policy field and analyse whether this mix had diversified over time. To view the development in the waste policy field in context, it was also decided that general trends in instrument choice for environmental policy as a whole would be reviewed. Sweden and England were chosen as case study countries because there were similar premises for a depoliticisation trend to potentially take place.

While there were some methodological and data difficulties, it can still be concluded that *EPI diversity has increased considerably in both countries' waste policy mixes*. In terms of the *number of new instruments* added to the mix, the development was most dramatic in England, with a six-fold increase from 1995 to 2005. This high level of activity can be explained by new waste-related EU legislation and new national waste strategies, that had emphasised the need for dramatic improvement (see chapter 5). In Sweden, there was more activity in waste policy in the 1990s, and national strategising on waste policy was not as urgent since Sweden had progressed further up the waste hierarchy. It was also seen that the *use of different EPI types* had increased in both countries, suggesting that the respective governments had been at least partially successful with their quests for NEPIs in the waste policy field.

In both countries, the use of *economic instruments* for waste in particular has increased since the mid-1990s. While Sweden has tended to use more negative economic incentives, in England there have been more positive economic incentives in the form of subsidies and grants. In England, the use of softer instruments such as information and education schemes and VAs has also been higher. In neither country was there evidence of a pure market or planning approach, but both have mixed approaches. However, there seems to be a tendency in the UK to choose market approaches to solve a given problem in the most cost-effective way or to internalise a measured externality. In Sweden, on the other hand, economic instruments have been justified in terms of applying the 'polluter-pays principle' or providing dynamic incentives for change of the waste management system in a more environmentally friendly way.

This study shows that diversification has indeed taken place in practice and not just in policy rhetoric. It is a different question whether the *process of diversification has been fast or slow*, and the existing literature has not yet settled on a unanimous assessment of the process as 'evolutionary' or 'revolutionary'. Clearly, it depends on what criteria are used for the pace of policy change. Considering the six-fold increase of EPIs in waste policy in England in the last ten years, though, the process in the UK seems to be revolutionary while the change in Sweden has been more modest.

To what extent are these results generalisable to other environmental policy fields? It was seen that the results from the waste instrument mix study corresponded to the findings from the overview of instruments used within *environmental policy as a whole*. This overview showed that economic instruments are now used extensively in both countries. Sweden is a world-leading user of environmental taxes and charges, while the UK has started operating several TPSs. Neither country have used VAs to a great extent so far, though. Meanwhile, it was also clear that the adoption of such NEPIs has not meant that the use of traditional command-and-control regulation has decreased. Overall, it was argued that more research is needed on possible measurements of the use of other instrument types than economic instruments and VAs, which may be overrepresented in the current body of studies.

Based on the existing literature on EPI diversity, it is difficult to assess whether the apparent success of the Swedish and UK governments in translating the commitments to

EPI diversity into actual EPI choice outcomes is representative of broader trends in Europe or OECD countries. It was noted in chapter 4 that there is a lack of international data and studies of EPI diversity, something which future research could address.

Why has diversification of waste policy instruments been more dramatic in England compared to Sweden? It was not the purpose of chapter 6 to examine the *reasons of diversification*, but some factors in addition to the political will associated with the commitments to introducing more NEPIs may provide some explanation. First, considering the nature of the waste problem and overarching waste policy framework, it is clear that the magnitude of the challenge facing the UK in the 1990s to reduce its historical reliance on landfill meant that there was a *problem urgency* that may have emphasised the need for creativity, innovation and comprehensive policy packages. Sweden had in a more gradual way decreased the dependency on landfill and managed to increase recycling more quickly. Second, due to the historically poor waste performance of the UK, it is possible that the scrutiny of and *demands for accountability* on the UK government to take effective action was stronger than in Sweden. Arguably, the UK government has recently exhibited more of an ‘entrepreneurial’ approach to waste policy, with its efforts to demonstrate that the problem is being tackled in a range of complementary ways and with comprehensive policy packages. Finally, there may be a natural appreciation of many NEPIs in the UK as a consequence of the *national policy style*. Discretion, voluntarism and flexibility have been described as key features of the British policy style (see **Table 9** in chapter 3), and these features arguably match well with many economic instruments and various soft instruments (e.g. VAs, informative measures).

The review of both the overall and waste-specific EPI mixes suggests that so-called *NEPIs are not really ‘new’ anymore* (cf. Jordan, Wurzel et al. 2003b; EEA 2006; Naturvårdsverket & Energimyndigheten 2006). It appears that the 1990s was the prime era of innovation of EPIs and many of the innovations have since then had time to become institutionalised in the national EPI repertoires. The overview of general EPIs indeed suggested that the *adoption of NEPIs had slowed down*. In the waste policy field, there was agreement among many policy-makers and stakeholders in both countries that EPIs introduced to divert waste to treatment options higher up in the hierarchy now needed time to settle and take effect. It is very clear in both countries’ last waste policy strategies that the emphasis now should be on ensuring effective implementation and enforcement of existing EPIs and removing other policy barriers (such as cumbersome planning permission for new recycling facilities) (see Naturvårdsverket 2005; DEFRA 2006).

What, then, are the possible *new trends in EPI choice*, generally and in the municipal waste policy field specifically? Firstly, in the waste policy field, it was seen in chapters 5 and 6 that quantified and timed *targets* have been increasingly adopted. This target-setting trend can be seen in the wider context of New Public Management, which both countries in this study have been inspired by. Arguably, targets could be seen as EPIs in their own right, rather than as preconditions or reference points for instruments. Interviewees in this study indicated that policy targets were taken seriously and could give rise to behaviour changes without an associated regulatory or financial instrument mechanism.

Secondly, policy attention is increasingly turned towards *waste prevention*, both at national and EU level (see Commission of the European Communities 2005c; Naturvårdsverket 2005; DEFRA 2006). It is argued that waste management is now largely under control, and policy needs to focus on the fundamental problem; the ever-increasing generation of waste. This development has meant that the waste policy field is likely to be significantly re-framed or partly replaced by new concepts and paradigms such as Integrated Product Policy and Sustainable Production and Consumption.

Thirdly, an increasing focus on household consumption as the root problem of municipal waste means that individuals as *consumers may increasingly become target actors of EPIs*. Arguably, directly addressing individuals has been politically sensitive, but a momentum now seems to be building up to change lifestyles through public policy instruments. For example, in the UK, waste collection charges have been included as a lump sum in the council tax, but there is now experimentation with separate waste collection charges to make visible the costs and differentiation of rates that will provide incentives to decrease householders' waste generation. As was noted in the introduction, the idea to target consumers is also being increasingly discussed within the climate policy arena, with instrument proposals such as personal carbon allowances. The Environment Agency in the UK recently commissioned work on behavioural economics, to see how such lessons could be incorporated into environmental policy instruments, e.g. use of rewards rather than punishments, written commitments (VAs), and use of ambassadors (Environment Agency 2005).

Finally, as discussed in chapter 2, there is an increasing interest in the policy-maker community to study *EPI mixes*, mainly from a technical point of view (see OECD 2003d; EEA 2005). It has been stated that synergies must be better captured and interaction effects between instruments better understood. A prerequisite for such studies is better data on EPIs in use, an issue that was discussed in chapter 4.

9.3 Coerciveness – increasing or decreasing?

In addition to examining the diversity of the municipal waste instrument mixes, it was argued that an analysis of the development of coerciveness of the mixes was necessary to understand how public policy instruments had increasingly or decreasingly intervened in private behaviour. Much of the existing EPI literature is concerned with technical properties of EPIs, rather than what level of intrusion they imply and what assumptions regarding human behaviour they are based on (see Schneider and Ingram 1990). Had the quest for NEPIs been associated with primarily 'softer' or 'harder' instruments?

The analysis of the pattern of coerciveness of the waste instrument mixes over the last decade in chapter 6 showed that *Sweden has adopted rather consistently coercive instruments*. While starting out with relatively coercive instruments, some more soft instruments were later adopted. This result contradicted the pattern predicted by Doern and Phidd, namely that liberal democratic governments would start with minimal-coercion instruments and then move up the scale only as necessary to address a given problem. The *England mix, on the other hand, was characterised by a give-and-take pattern*, as predicted by van der Doelen's theory on instrument choice.

A possible reason for these patterns is that the mixes resulted from 'kitchen-sink approaches' with no clear government strategy (see Gunningham and Grabosky 1998), but this seems unrealistic since policy actors would hardly accept that consideration of a new EPI is divorced from the existing policy instruments in a given field. Another possibility is that *problem urgency* determines the development of the mix over time. As explained in chapter 5, changes in line with the waste hierarchy had been made earlier in Sweden, which may have caused the gradual ‘softening’ of instruments. In the UK, the problem urgency increased as legal targets were imposed to limit landfill. Accordingly, the UK had to adopt ‘harder’ instruments’, but they were complemented by ‘softer’ instruments to facilitate the transition. Finally, one could look for *political and ideological factors* explaining the acceptance of less or more coercive instruments. Doern and Phidd (1983) viewed ideology as the key determinant. Considering that UK governments have been more liberal than Swedish government over the last decade, the lower degree of coerciveness found in the England mix is not surprising. Van der Doelen (1998) stated that it was the need to legitimise ‘repressive’ instruments that led to the adoption of ‘stimulative’ instruments. It was found in the EPI process case studies (chapters 7 and 8) that Sweden is a more ‘statist’ country and evidently the government could introduce a new instrument despite strong stakeholder opposition. This raises the question whether the general need to legitimise policy instruments is lower in Sweden than in the UK, hence the less common use of ‘stimulative’ instruments.

In chapter 6, the analysis of coerciveness was made with reference to the *types* of EPIs chosen, with the assumption that a certain instrument type was automatically associated with a certain level of coerciveness. However, it was argued that the actual coerciveness of an EPI is not necessarily dependent on the type of instrument, but its *specific design features* (see Macdonald 2001; Daugbjerg 1999). Therefore, it was argued that future analysis of coerciveness would benefit from and become more accurate by a closer examination of design aspects of instruments. Furthermore, it was suggested that the concept of coerciveness needs to be unpacked and five possible indicators were suggested for this purpose: the nature of the target group; universal application or individual adaptation; net transfer of resources from government to target actors and vice versa; level of ambition; and level of precision. The existence of credible enforcement and sanctions is also likely to affect the perceived coerciveness of an instrument.

9.3 Procedural rationality in the EPI choice process

Having examined the EPI mixes, the thesis then moved on to analyse the process of EPI choice and two case studies were made for this purpose; the landfill allowance trading scheme (LATS) in England and the tax on waste incineration in Sweden. The first question was whether EPI choice processes are characterised by procedural rationality or not, and whether they in this regard can be considered depoliticised. A second question is whether the level of procedural rationality contributed to the choice of a NEPI as opposed to traditional regulation. Below, the degree of procedural rationality *per se* will first be discussed, since this is a general policy-making ideal that both countries strive towards.

The case studies of the two recently adopted NEPIs in Sweden and England respectively showed that *in the England case there was a high level of procedural rationality*, whereas the *Swedish case was characterised by a low level of procedural rationality*. These case studies thus suggests that the UK is more successful than Sweden in translating this ideal into policy practice. There may thus be reason to debunk the myth of Swedish policy-making as 'rationalistic' (see Ruin 1982) and UK policy-making as more 'informal' and 'pragmatic' (Carter and Lowe 1998) (see **Table 9** in chapter 3) through further research based on a wider set of cases.

However, it was found that a number of *case-specific circumstances* affected the result and, consequently, the generalisability of the findings. First, the England EPI choice process was a response to EU legislation, which in this case meant that the *problem definition and policy targets were externally imposed* upon the UK policy-making community. This meant that important building blocks for a continued process of procedural rationality had already been laid and that actors were united in wishing to solve the problem as rationally and efficiently as possible. This raises the question of whether EU legislation in general has the effect of increasing the degree of procedural rationality in the search for EPIs at the national level, as compared with domestically initiated and more 'proactive' processes such as that in the Swedish case study.

Second, it was found that the *high-cost nature of the problem* addressed by the England case study instrument motivated more rigorous policy analysis, compared with the

Swedish case where the problem at hand had more of a fine-tuning character. Thirdly, it was argued that the *policy problem complexity* matters for the ease with which procedural rationality can be sought and achieved. The waste hierarchy, in which waste management options are *relatively* better or worse than others, may be more problematic than in cases where policy objectives are more absolute in nature. Furthermore, in the Swedish case study, both the energy and waste management systems had to be considered jointly, which meant that there were complicated effects and assumptions to consider in the process.

Finally, it was argued that *RIA can serve as an effective vehicle for procedural rationality* throughout an EPI choice process, since it forces explicit determination of each decision-making stage: problem definition, policy objectives, alternative instruments, impact assessment, and final decision. It was found that the UK system of RIAs facilitated this role, while the lack of standardised formats for RIA in Sweden meant that a transparent comparative assessments of several EPIs was not obligatory.

There may also be underlying reasons why procedural rationality was found to be lower in the Swedish case. An EPI choice process characterised by strong procedural rationality may primarily be a result of *a need to increase the legitimacy of policy decisions*, rather than the ambition to make informed decisions. The argument that various routines, such as problem analysis and RIA, is part of a *post-hoc* rationalisation effort only, has been made by several commentators (Howlett 2000b; Rydin 2003). Howlett has even argued that these kinds of routines and measures should be seen as instruments in their own right, namely as ‘procedural instruments’, complementary to ‘substantive instruments’ (such as taxes, subsidies, regulations). Whereas the latter directly affect policy outcomes, the former are designed to “indirectly affect outcomes through the manipulation of policy processes” (p. 413), more specifically the relationships among actors involved. Procedural instruments include education, training, institution creation, formal evaluations, etc. The reason such instruments have emerged as a complement to traditional substantive instruments is the ‘hollowing-out’ and loss of autonomy of the state. His thesis is that “when a serious loss of legitimacy or trust occurs, the subject of political conflict often shifts from the actual substantive content of government actions towards a critique of the processes by which those actions are determined” (p. 422). The existence of such procedural instruments was not examined in the analysis of the waste policy mix summarised above, but is an issue for future research.

Possibly, there is a *greater need to legitimise policy decisions in the UK compared to Sweden*. The policy style of Sweden has been described as traditionally rationalistic with a belief in technical problem-solving based on comprehensive knowledge (Richardson, Gustafsson et al. 1982) and the level of trust in the political system may be comparatively high (see Jagers and Hammar forthcoming). Although the stakeholders in the Swedish case study called for a more rational process with greater consideration of alternative instruments and clearer policy objectives, it was not seen as a systemic problem but rather specific to this case. In the UK, the need to gain legitimacy by demonstrating a rational procedure of EPI choice is possibly higher. However, further research is needed to satisfactorily examine this question.

Accepting that the observed (high or low) degree of procedural rationality may be a result of a *post-hoc* rationalisation effort rather than a genuine effort to systematise and inform EPI choice, does this mean that the ideal cannot and should not be pursued? There are at least three reasons why it could still be considered a relevant ideal. Firstly, it is, obviously, an ideal rather than a realistic description of policy-making practice. Brunsson (2006 p. 11) has recognised the importance of ideals, even though they may not necessarily be achieved in practice:

> "Western culture is a culture of hope. We can easily live in two worlds simultaneously: the world as we believe it to be, and the world as we think it ought to be. These worlds need not have much in common. But the fact that 'is' and 'ought to' often diverge does not cause us to abandon faith in one or the other. We tend to behave in the practical world on its terms, without abandoning our ideals. Even when we notice discrepancies between the two worlds, we tend to reconcile them through hope: We hope to bring about agreement between the way things are and the way things ought to be."

Second, it may be misleading to think of 'informative' procedural rationality and 'legitimising' *post-hoc* rationalisation as mutually exclusive. It could be a form of dialectical relationship, in that the policy-maker knows beforehand that he/she must justify a decision in a particular way. The actions taken to that end may modify parts of the decision while not completely altering it. Introducing a culture of demonstration of rationality of decisions *post-hoc* may also gradually, over time spill over in more 'informative' procedural rationality.

Third, while political factors beyond the control of the policy-maker conducting a particular EPI choice process may determine the choice outcome, introducing elements of procedural rationality at the very least forces a degree of transparency. Political factors may be completely legitimate and publicly accepted, and we may not want overly proceduralised and professionalised processes where bureaucrats effectively make key decisions rather than elected politicians. However, a minimum level of transparency is a basic democratic value. One could think of procedural rationality as creating ‘windows’ for political factors to shape the EPI choice, e.g. in setting choice criteria or formulating policy objectives. Procedural rationality then imposes a certain structure to the process in which policy argumentation by the stakeholders involved can be more easily scrutinised and disputed.

We can thus conclude that these country cases provided contradictory results as to whether there has been a (successful) resurgence of procedural rationality in practice. Generalising these results would require further case studies. It can be noted, however, that studies that have examined the general practice of RIA within the environmental policy field seem to suggest that the UK is increasingly achieving a good performance while Swedish practice is still weak (see Persson 2003; National Audit Office 2006b; Russel and Jordan 2007). It remains to be seen if the new RIA regulations that are currently being prepared in Sweden will improve current practice.

9.4 Has EPI choice become depoliticised?

Considering the empirical results presented in the previous chapters, can we observe a depoliticisation of EPI choice in Sweden and the UK? If we take the diversification of the EPI mix and increasing use of NEPIs as an outcome indicator, then the answer is yes. If we take the ‘proceduralisation’ of the choice process as a second indicator, we can observe a higher degree of depoliticisation in the England case than in the Swedish case.

However, considering that a ‘new’ EPI was chosen in both cases, is procedural rationality a necessary and sufficient condition for the consideration and adoption of NEPIs? With regards to *consideration* of EPIs (including NEPIs), the England case showed that the RIA process contributed to the identification of alternative instruments. In the Swedish case, the decision to study alternative instruments (in a rather limited way) instead resulted from party-political negotiations. With regards to eventual *adoption* of NEPIs, the Swedish case suggests that procedural raitonality is neither necessary nor sufficient. The evidence from the England case does not support an opposing conclusion, since we do not have a counterfactual to compare with. However, it is reasonable to conclude that the high procedural rationality facilitated and contributed to the final choice, if not a necessary condition. The policy process should be more conducive to ideas on new forms of EPIs when it is open, formalised and transparent. It should again be emphasised, though, that this study is exploratory in nature and that more extensive case studies are needed for more conclusive results.

What factors then made the choice processes politicised, and what factors explain the actual choices made, i.e. a tax and a TPS? Indeed, was it at all a free 'choice' of EPI in these cases? For this analysis, the framework of *political and institutional factors at the macro, meso and micro levels* was used. The theories included in this framework provided both perspectives for understanding the process and predicted certain EPI choices. Importantly, these levels should be seen as nested and theories concerned with factors at different levels are complementary rather than rival theories.

In the *England case*, several political and institutional factors were found to work 'with the grain' of a process characterised by procedural rationality and an active search for a 'new' EPI. First, the *legal framework*, in this case the EU Landfill Directive, stipulated that an instrument had to be developed, which urged an analysis of effective and cost-effective instruments. The EU Landfill Directive also contributed to a *paradigmatic change of waste policy*, where new ideas on how to achieve aims and targets had to be considered. Secondly, New Labour had made a general commitment to use economic instruments to a wider extent, hence encouraging the choice of a TPS. Furthermore, the choice of a TPS as an economic instrument was not constrained by party-politics or ideological cleavages. Finally, at the micro-level, it was seen that the costliness and coerciveness of a TPS was more easily accepted since the target actors (i.e. local authorities) could be *financially compensated*. This compensation may also explain why the adoption of the instrument went so smoothly and the appearance of procedural rationality was so high.

In the *Swedish case*, the process was highly politicised and there were obstacles to procedural rationality. The most important driver on the choice of a tax in this case was the *party-politics* involved. Although portrayed as an instrument to achieve increased recycling and less incineration, it was found that the informal objective was to find a new tax base to the *green tax shift*. The Green and Left Parties, which the Social Democratic relied on for parliamentary majority, were instrumental in introducing the tax. Their bargaining power led to the *a priori* commitment to a tax instrument, which precluded the need for procedural rationality. It is likely that the process would have taken longer and possibly ended up in another instrument choice (e.g. a better enforced producer responsibility for plastics), or no instrument at all, if there had not been a party-political pressure. *Opposing ideas and perspectives on basic waste policy principles* among key stakeholder groups meant that a commonly agreed problem definition was impossible, and thus hindered procedural rationality. The fact that the government could decide on a tax in the face of massive critique could be explained by the *statist political culture* in Sweden, which was reflected in the lack of power of target groups at the micro-level to realise their first preference for the no-action alternative or a positive economic instrument. Finally, it was seen that environmental taxes fitted with the *policy style and institutional arrangements* in Sweden.

The macro, meso and micro level analytical framework was thus helpful for understanding why the choices of a TPS and a tax respectively were made. As described in chapter 4, the purpose of these two case studies was not to rigorously test the alternative instrument choice theories as rival theories, but rather to support the interpretation of the cases. The results suggest, though, that in particular the role of the national policy style and political culture, party politics and micro-level bargaining between actors could be important to clarify and examine further in future research.

9.5 Beyond the quest for NEPIs

As the adoption of NEPIs seems to be slowing down, questions arise as to why this is happening and if it really matters? Does it mean that the momentum to innovate in environmental policy has weakened? Based on the empirical findings presented in this thesis, two critical issues for the future study of EPI choice have been identified.

First, the recent shift from focusing on single kinds of EPIs to recognising the need for comprehensive mixes has almost led to a situation where *high instrument diversity is seen as an end in itself*. Arguably, many key environmental policy statements in the 1990s viewed a wide repertoire of EPIs and high use of NEPIs as an inherently good feature of environmental governance, rather than for the potential gains in environmental effectiveness and economic efficiency. A possible risk with a ’diversification for diversification’s sake’ approach is that policy-makers may not tackle the root causes of problems and the failures of existing instruments, if they focus on introducing new ones, and thereby create ‘policy messes’ (Sorrell and Sijm 2003) or ‘policy congestion’ (Majone 1989). As described above, a very tentative observation is that a more entrepreneurial approach to environmental policy may be emerging, where policy-makers are keen to demonstrate that comprehensive policy packages and strategies are developed and that ‘things are getting done’.

On the other hand, there also seems to be a few signs that a more self-critical approach is taken towards environmental policy. First, there has been an increasing emphasis on ensuring the effective implementation and enforcement of EPIs already in use (Naturvårdsverket 2005; DEFRA 2006; Commission of the European Communities 2005c). Second, there have been calls for more *ex post* evaluation of EPIs, to better understand their *actual* effectiveness and comparative performance (EEA 2001b). As described by Macdonald (2001), there may be significant variations of effectiveness *within* a certain EPI category depending on the level of enforcement. In a similar vein, Daugbjerg (1999) has argued that design features (e.g. level of a tax rate) determine the actual impact of an instrument, not the generic type it belongs to. It may be incorrect to assume that adding new types of EPIs to a mix will automatically enhance its effectiveness.

A second question relates the process of choosing among alternative EPI types, including the so-called ‘new’ EPIs. Does it really matter if it is depolititicised or not? So far, the NEPI agenda has often focused on instrument *types*, i.e. the six general categories of instruments listed in chapter 1. However, one could argue that the *development of a new EPI involves three decisions*; first, whether to publicly intervene in private behaviour or not; second, which EPI type to choose; and third, decisions on design features of the instruments. Possibly, the second decision on EPI type may have become depoliticised over time, as described in the introduction to this thesis.

The first kind of decision is arguably still perceived as highly ideological. In the case studies, it was clear that the bulk of the debates centred upon the problem definition and whether it justified public policy intervention. A non-negligible issue for target actors could be the total scope of their behaviour that is somehow regulated by public policy, as opposed to how they are regulated.

The third kind of decision appears to often be highly political, in terms of the redistributive implications that can arise from specific designs. Daugbjerg (1999) has argued that it is the cost-benefit ratio resulting from an instrument's design that really matters and where controversy arises, rather than type of EPI. In the case studies on the English LATS and the Swedish incineration tax, it was clear that the design features generated much discussion and bargaining, while the agreement on the type of EPI came quickly in the case of the LATS and was sidelined in the case of the incineration tax.

To conclude, by the mid-2000s there may now be a situation where the NEPI quest formulated in the 1990s has served its purpose, in diversifying EPI mixes and removing old ideological biases for or against certain types of EPIs.

9.6 Policy implications and issues for future research

Based on the theoretical review and empirical results presented in this thesis, some concrete *policy implications and recommendations* can be identified.

- First, in order to get a better overview of the use of different EPIs in different policy fields, policy-makers should *systematically and regularly record the instruments* in a coordinated way. In this way, the cumulative instrument mixes in use in different fields would become more transparent to both stakeholders and students of environmental policy.
- Second, the inquiry into waste policy instruments in use revealed that there was limited follow-up and evaluation of the effectiveness of EPIs used. *More ex post evaluation* would not only facilitate the improvement of the performance of the

individual instrument, but would also contribute to the body of knowledge around its particular category of instruments in comparison to other categories.

- Third, the records of EPIs in use and evaluations of their effectiveness should be more systematically used in the initial phases of consideration of new instruments (or removal of existing ones). An *analysis of the existing mix* is a prerequisite for identifying positive synergy effects and potentially negative interaction effects with new instruments, as well as for avoiding creating situations of 'policy congestion'. More academic work on instrument mixes may support the development of methodologies for mapping out mixes in a useful way.
- Finally, since it was found that *formal and standarised RIA* serves as a vehicle for increasing the procedural rationality of the process and for enhancing transparency, further institutionalisation of this tool would improve EPI choice processes.

A range of issues in need of further research also arose during the course of this research. As described in chapter 4, there were several methodological and data-related difficulties that had to be overcome. One of the major problems was the lack of inventories of EPIs in use, that covered all six EPI categories and were complete in terms of international coverage and up-to-date information. Therefore, such inventories had to be prepared within this research (see Appendices) and took time from more advanced analysis of them. Upon reflection, other research designs could have been considered, that either included a higher number of and more different country cases, comparison of the waste policy field with some other environmental policy field, and/or a higher number of EPI choice processes as case studies. Such research designs would have increased the generalisability. For this thesis, a more exploratory approach was chosen in the end.

Based on the empirical findings and theoretical issues explored in this thesis, future research in any of the following directions would be relevant and useful:

- First, the study of EPI diversity could be expanded to include other European countries (e.g. Southern European countries) and other environmental policy fields, in order to enable more comparative analysis.

- Second, by unpacking the concept of coerciveness and using more specific indicators such as those suggested above, a more fine-grained study of the pattern of coerciveness in a EPI mix could be performed. This would also make cross-country comparisons of coerciveness more accurate. Furthermore, it would be interesting to understand the perception of coerciveness of instruments among policy actors to a greater extent.
- Third, the study of EPI choice as a process would benefit from comparison with more cases, in order to enable more generalisable results in relation to the degree of procedural rationality and the significance of various political and institutional factors.
- Finally, given the possibly increasing need to legitimise environmental policy towards the public, it would be relevant to examine whether Howlett's (2000b) notion of 'procedural instruments' (see above) can be supported by empirical evidence.

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APPENDICES

APPENDIX I

List of case study written material

Sweden – Tax on waste incineration

Bills, regulations and parliamentary records

- Regeringens proposition (2003). 2002/03:117. Ett samhälle med giftfria och resurssnåla kretslopp. Stockholm, Regeringskansliet.
- Miljö- och Jordbruksutskottet (2003). Utskottsbetänkande 2003/04:MJU4. Ett samhälle med giftfria och resurssnåla kretslopp. Stockholm, Sveriges Riksdag.
- Regeringens proposition (2005). 2005/06:1. Budgetproposition för år 2006. Stockholm, Regeringskansliet.
- Regeringens proposition (2006). 2005/06:125. Beskattning av visst hushållsavfall som förbränns, m.m. Stockholm, Regeringskansliet.
- Skatteutskottet (2006). Utskottsbetänkande 2005/06:SkU33. Beskattning av visst hushållsavfall som förbränns m.m. (prop. 2005/06:125 delvis). Stockholm, Sveriges Riksdag.
- *Lag (2006:925) om ändring i lagen (1994:1776) om skatt på energi.*

Consultation papers/Commission of inquiry reports

- Statens Offentliga Utredningar (1990). SOU 1990:59. Sätt värde på miljön! Miljöavgifter och andra ekonomiska styrmedel. Stockholm, Regeringskansliet.
- Statens Offentliga Utredningar (1994). SOU 1994:114. Avfallsfri framtid. Stockholm, Regeringskansliet.
- Statens Offentliga Utredningar (1996). SOU 1996:139. Skatt på avfall. Stockholm, Regeringskansliet.
- Ministry of Finance (2001). Kommittédirektiv Fi 2001:13. Vissa avfallsskattefrågor.
- Statens Offentliga Utredningar (2002). SOU 2002:9. Skatt på avfall idag - och i framtiden. Stockholm, Regeringskansliet.
- Naturvårdsverket (2002). Ett ekologiskt omhändertagande av avfall. Rapport 5177. Stockholm, Naturvårdsverket.
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- Statens Offentliga Utredningar (2005). SOU 2005:23. En BRASkatt? Beskattning av avfall som förbränns. Stockholm, Regeringskansliet. (Including different versions of draft chapters)

Regulatory Impact Assessments

- Ekonomistyrningsverket (2003). Checklista för konsekvensutredningar. 2003.
- (RIAs are integrated into the commission of inquiry reports and not published separately)

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- Profu (2003). Konsekvensanalyser av olika framtida styrmedels- och skattemodeller för det svenska energisystemet - ett samarbetsprojekt med Svensk Energi, Svenska Fjärrvärmeföreningen, Svenska Bioenergiföreningen och Svenska Gasföreningen. Huvudrapport. Mars 2003. Göteborg, Profu.
- Profu (2005). Skatt på förbränning av avfall – En konsekvensanalys 2005-02-28.
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Consultation responses and stakeholder position papers

Summaries of consultation outcomes

- Ministry of Finance (2002). Fi 2002/589. Sammanställning av remissyttranden över betänkandet Skatt på avfall idag - och i framtiden (SOU 2002:9). Stockholm, Regeringskansliet.
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Left Party (Vänsterpartiet)

- Vänsterpartiet (2005). Pressmeddelande 2005-09-15: KLIMP räddas kvar och skatt på avfallsförbränning införs!
- Vänsterpartiet (2005). Faktablad: Avfallshantering.

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APPENDIX II

List of interviewees

SWEDEN

Interviewees marked with (*) participated in the 2003 BRAS Inquiry as committee secretaries, advisers (ministry officials) or experts (government authority officials and external stakeholders).

CENTRAL GOVERNMENT

2003 BRAS Inquiry Committee Secretariat
*Hanna Werth, Committee Secretary, Stockholm, 16 September 2005
*Henrik Hammar, Committee Secretary, Stockholm, 23 September 2005

Ministry of Finance
*Susanne Åkerfeldt, Senior Adviser, Stockholm, 14 September 2005
*Magnus Schulzberg, Desk Officer, Stockholm, 14 September 2005

Ministry of Industry, Employment and Communication
*Sven-Olov Hansson, Deputy Director, Stockholm, 7 September 2005

Ministry of Sustainable Development (previously Ministry of the Environment)
*Viktoria Ljung, Deputy Director, Stockholm, 14 September 2005

Swedish Environmental Protection Agency
*Björn Södermark, Deputy Director, Stockholm, 6 September 2005

Swedish National Tax Board
*Kristina Dahlqvist, Desk Officer, Telephone interview, 27 September 2005

LOCAL GOVERNMENT

The Swedish Association of Local Authorities and Regions
*Ronnie Peterson, Tax Expert, Stockholm, 27 September 2005

WASTE MANAGEMENT INDUSTRY

The Swedish Association of Waste Management (members include local authorities, municipal and regional companies, manufacturers, consultants and contractors)
*Weine Wiqvist, Managing Director, Malmö, 26 September 2005

The Swedish Recycling Industries' Association
*Annika Helker Lundström, Director, Telephone interview, 27 September 2005

Tekniska Verken i Linköping AB (regional utility company)
*Ingvar Carlsson, Vice President, Linköping, 12 September 2005

MANUFACTURING INDUSTRY

Confederation of Swedish Enterprise (and Plast- och Kemiföretagen)
*Anders Norman, Desk Officer Environment, Stockholm, 15 September 2005

ENVIRONMENTAL NON-GOVERNMENTAL ORGANISATION

The Swedish Society for Nature Conservation
Mona Blomdin Persson, Head of Environmental Protection, Stockholm, 19 September 2005

OTHER EXPERTS

*Göran Finnveden, Associate Professor, Department of Industrial Environmental Protection, Royal Institute of Technology and Swedish Defense Research Agency (FOI), Stockholm, 28 September 2005

*Roy Resare, one-man committee of 2001 Waste Tax Inquiry and former Green Party MP, Sundsvall, 22 September 2005

ENGLAND

CENTRAL GOVERNMENT

Department of Environment, Food and Rural Affairs (DEFRA)
Martin Cox, Waste Strategy Review Team Leader, London, 10 May 2005
Thomas McGowan, Policy Adviser, Waste Strategy Review Team, London, 10 May 2005
Andy Doran, Head of Local Authority Waste Performance, London, 11 November 2005
Amy Glover, ex-Landfill Policy Team, London, 18 November 2005
David Wood, Economist, London, 18 November 2005

Environment Agency for England and Wales (EA)
Liz Parkes, Head of Waste Regulation, London, 30 November 2005
Fran Lowe, Project Manager LATS, London, 30 November 2005

LOCAL GOVERNMENT

Local Government Association (LGA)
Alice Roberts, Senior Project Officer, Environment, telephone interview 2 February 2006

Local Authority Recycling Advisory Committee (LARAC)
Andrew Craig, Policy Officer, North East (also Waste Management Development Officer, Tees Valley Joint Strategy Unit), telephone interview 27 January 2006

WASTE MANAGEMENT INDUSTRY

Environmental Services Association, ESA
Jacob Hayler, Economist, London, 1 February 2006

Biffa Waste Services
Peter Jones, Director External Relations, London, 18 November 2005

PROFESSIONAL ASSOCIATION

Chartered Institute of Waste Management (CIWM)

Tina Benfield, Technical Officer, Northampton, 23 February 2006

ENVIRONMENTAL NON-GOVERNMENTAL ORGANISATION

Green Alliance
Ben Shaw, Principal Policy Adviser, London, 27 February 2006

CONSULTANTS

Eunomia
Dominic Hogg, Director (previously with ECOTEC consultants), Bristol, 30 January 2006

Enviros
Guy Turner, Director of Climate Change Policy and Strategy (previously also worked on waste policy), London, 6 February 2006

APPENDIX III

Interview topic guide

Adapted after the interviewee and his/her organisation's role in the EPI choice process and case study (Sweden or England).

Name
Organisation
Position

BACKGROUND

- Which organization do you represent and what do you do?
- What is your current role in policy-making?
- Over the last 5-10 years what has your role been?

EPI CHOICE PROCESS

- How did the problem arise and appear on the policy agenda?
- Was there a decision on a policy objective first? What was the objective? Were there conflicting objectives to consider?
- What kind of key criteria (e.g. economic efficiency, environmental effectiveness, distributional equity, legitimacy and accountability, or technical and administrative feasibility) were specified at the start of the process? Did they change subsequently?
- Who took the initiative to start the instrument choice process?
- Who was involved in the process?
- How much consultation with or participation of external stakeholders was there?
- Which instruments were considered at the outset (if more than one)?
- Were there significant constraints to considering alternative instruments, e.g.
    - legal constraints (e.g. EU regulation, national legislation)
    - technological constraints
    - lack of time and resources to analyse alternative EPIs
    - previous political commitments to a certain course of action
- Was an Impact Assessment made of alternative EPI types and/or designs? If so, was it adequate in scope and quality? Was it influential on the choice outcome?
- What were the most critical or controversial issues when choosing and designing the instrument?
- Why was the chosen instrument preferred in the end?
- What kind of instrument did you/your organization prefer and why?

OTHER

- What do you think of future instrument trends in the municipal waste policy field?
- Do you see a pattern in this mix, i) a balance between stimulative and repressive instruments ii) gradually more coercive instruments being chosen?
- Why do you think this pattern has emerged?

APPENDIX IV

EPIs used in Sweden and the UK

ENVIRONMENTALLY RELATED TAXES AND CHARGES

Taxes	Environmental policy field**	Year introduced (and discontinued)	Revenue, million SEK or GBP in 2005 (or latest year available)
Sweden			
TAXES			
*Tax on pesticides	Water pollution	1984	78
*Tax on artificial fertiliser	Water pollution	1984	329
# - *Sales tax on motor vehicles (differentiated by environmental classification)	Air pollution; Climate change; Energy efficiency; Transport	1993-2002	(2002: 15)
*Motor vehicle tax (differentiated by environmental classification)	Air pollution; Climate change; Energy efficiency; Transport	2002	10,249
# - *Kilometre tax	Air pollution; Climate change; Energy efficiency; Transport	?-1994	(1994: 10)
Fuel taxation differentiation			
# - Leaded and unleaded petrol	*Air pollution; Transport*	*1986-1995*	-
Diesel	*Air pollution; Transport*	*1991*	-
Petrol sulphur content etc.	*Air pollution; Transport*	*1994*	-
# - *Tax on domestic aviation (emissions of hydrocarbons and nitrogen oxides)	Air pollution; Climate change; Transport	1989-1996	(1996: 117)
*Energy tax	Air pollution; Climate change; Energy efficiency; Transport	1991	37,884
*CO2 tax	Air pollution; Climate change; Energy efficiency; Transport	1991	25,535
*Sulphur tax	Air pollution; Climate change; Energy efficiency; Transport	1991	75
# - *Hydropower tax	Energy efficiency	?-1996	(1996: 1,520)
# - *Special tax on electrical power from nuclear power	Air pollution; Climate change; Energy efficiency	1983-2000	(1999: 1,545)
*Tax on thermal effect in nuclear reactors	Water pollution	2000	1,794
*Natural gravel tax	Land contamination; Natural resource management; Land management	1996	200
*Landfill tax	Waste management	2000	736
Tax on waste incineration	Waste management	2006	-
CHARGES			
Water pollution fee (oil discharges at sea)	Water pollution	1984	0.1
Charge on NOx emissions	Air pollution	1992	55.9
# - Noise related landing charges for	Noise	1994-1998	0.1

airplanes			
Environmental differentiation of aviation charges (noise, emissions of hydrocarbons and NOx)	Air pollution; Climate change	1998	-
Differential harbour and shipping lane dues (emissions of sulphur and CO2)	Air pollution; Climate change	-	-
Road charges for heavy vehicles, differentiated by exhaust classes	Air pollution; Climate change; Transport	1998	54.7
Excavation charge	Natural resource management; Land management	1984	-
Licence fee for exploitation of peat	Natural resource management; Land management	1985	-
Hunting fee	Natural resource management; Land management	1995	1.6
Battery fee	Waste management	1987	13.2
Fee on imported aluminium cans	Waste management	1984	7.7
*Charge on nuclear waste	Waste management	1982	690
Rural settlement and fisheries charge	Natural resource management; Land management	-	-
UK			
TAXES			
*Air passenger duty	Air pollution; Noise; Energy efficiency; Transport	1994	909
*Duty on hydrocarbon oils (differentiated rates for unleaded petrol, leaded petrol, ultra low sulphur petrol, diesel, ultra low sulphur diesel)	Air pollution; Climate change; Energy efficiency; Transport	(unleaded petrol differentiation 1987-2001; fuel duty escalator 1993-1999)	23,346
*' ***VAT on hydrocarbon duty (VAT on oil including duty)	Air pollution; Climate change; Energy efficiency; Transport	-	4,086
# - *Fossil fuel levy	Air pollution; Climate change; Energy efficiency	1989-2002	(2002: 32)
# - *' ***Gas levy	Air pollution; Climate change; Energy efficiency	1981-1998	(1998: 32)
*Climate change levy	Air pollution; Climate change; Energy efficiency	2001	744
# - *' ***Hydro-benefit	Energy efficiency	?-2004	10
*Vehicle excise duty (differentiated based on CO2 emissions)	Air pollution; Climate change; Energy efficiency; Transport	(env. differentiation from) 1998	4,809
*Aggregates levy	Land contamination; Natural resource management; Land management	2002	333
*Landfill tax	Waste management	1996	735
CHARGES			
Water abstraction licence charge	Natural resource management	-	(1998: 83.5)

- Indicates that the instrument has been discontinued.
*Taxes/charges that are reported in the official statistics on environmental taxation in the national environmental accounts, see National Statistics (2006) and SCB (2005).
**Classification of environmental policy field taken from OECD database on environmentally related taxes and charges.
***These UK taxes will not be classified as environmental taxes after the Spring 2006 edition of the National Environmental Accounts (Gazley 2006).

Note: The definition of environmental taxes and charges are not consistent across the sources used here, see comment in sections 4.2.3 and 6.2.1.
Sources: EEA (2001); Jordan, Wurzel et al. (2003e); Naturvårdsverket (2003); OECD (2003b); OECD (2003c); EEA (2005); SCB (2005); National Statistics (2006, p. 42, table 3.1); Naturvårdsverket & Energimyndigheten (2006) ; Gazley 2006.

TRADABLE PERMIT SYSTEMS

Tradeable permit systems	**Environmental policy field**	**Year introduced**
Sweden		
Tradable renewable energy certificates ('Green electricity certificates')	Climate change	2003
Emissions trading scheme (GHG)	Climate change	2005
UK		
Emissions trading scheme (GHG) (national scheme in 2002, succeeded by EU trading scheme in 2005)	Climate change	2002
Tradable renewable energy certificates (Renewables Obligation)	Climate change	2002
Packaging recycling notes	Waste management	1998
Landfill allowance trading scheme	Waste management	2006
Water abstraction license trading	Water	2003

Sources: Naturvårdsverket (2003); OECD (2003c).

DEPOSIT-REFUND SYSTEMS

Deposit-refund systems	**Environmental policy field**	**Year introduced**
Sweden		
Deposit-refund system for motor vehicles	Waste management	1975
Deposit-refund system for aluminium cans	Waste management	1982
Deposit-refund system for glass and PET bottles	Waste management	1991
UK		
-	-	-

Sources: Naturvårdsverket (2003); OECD (2003c); Naturvårdsverket & Energimyndigheten (2006).

ENVIRONMENTAL SUBSIDIES AND GRANTS

Environmentally motivated subsidies	Environmental policy field	Year introduced	Expenditure
Sweden			
Local investment programmes	General	1997-2002	6,210 million SEK 1997-2002
Local climate investment programmes	Climate change	2002	840 million SEK in 2002-2004
Tax subsidy for biofuels (ethanol, RME, biogas)	Climate change		
Grants for reducing hazardous emissions from domestic fuel tanks	Air pollution		
Grants for disposal of oil waste from ships	Water pollution; Waste management		
Environmental grants for measures to combat nitrogen leakage	Water pollution		
Grants, soft loans for biofuels.	Natural resource management; Energy efficiency		
Subsidy for biodiversity, habitats, landscape and cultural heritage	Natural resource management; Land management		
Subsidy for forestry	Natural resource management		
Subsidy for lake and water course liming	Natural resource management		
Environmental grants for creation and management of wetlands in agricultural areas	Natural resource management; Land management		
Environmental grants for creation of protection zones	Natural resource management; Land management		
Grants for investment in forests to improve their ecological value	Natural resource management; Land management		
Environmental grants for conservation of valuable natural environments and cultural heritage in agricultural areas	Natural resource management; Land management		
Environmental grants for conservation of grazing pastures and scythed meadows	Natural resource management; Land management		
Environmental grants for conservation of endangered breeds of domestic animals	Natural resource management; Land management		
Environmental grants for a varied and open agricultural landscape	Natural resource management; Land management		
Environmental grants for organic production	Natural resource management; Land management		
Environmental grants for environmentally friendly cultivation of kidney beans on Öland	Natural resource management; Land management		
Environmental grants for environmentally friendly cultivation of sugar beet on Gotland	Natural resource management; Land management		
Investment grants for agricultural, horticultural and reindeer herding enterprises	Natural resource management; Land management		
Adjustment and development in rural areas	Natural resource management; Land management		
Grants for management of reindeer enclosures	Natural resource management; Land management		
Grants for measures against radon in housing	Radiation		
Investment grants for ecological building	Building and construction		

UK			
Enhanced Capital Allowances for energy saving technologies	Energy efficiency	2001	-
Green Technology Challenge	Climate change; Energy efficiency	2002	-
Reserves Enhancement Scheme	Land management	1992	2003/04: 1.5 million GBP
# - Wildspace subsidies	Land management	2001-2006	
Grant for salt marshes	Natural resource management; Land management		
Grants for nitrate sensitive areas	Water pollution; Natural resource management; Land management		
Subsidy for farm waste treatment facilities	Natural resource management; Land management		
Subsidy for site conservation	Natural resource management; Land management		
Subsidy for wetlands	Natural resource management; Land management		
Subsidy for wildlife	Natural resource management		
Subsidy for woodlands	Natural resource management; Land management		
The England Rural Development Programme	Climate change; Natural resource management; Land management	2000	c. 230 million GBP per year 2000-2006
Country Stewardsship Scheme			
Environmentally Sensitive Areas Scheme			
Organic Farming Scheme			
Energy Crops Scheme			
Farm Woodland Premium Scheme			
Woodland Grant Scheme			
English Woodland Grant Scheme			
Hill Farm Allowance			

- Grants and subsidies that have been discontinued.
Sources: SCB (2006); Naturvårdsverket (2003); OECD (2003c); Naturvårdsverket & Energimyndigheten (2006).

VOLUNTARY APPROACHES

Note: Only negotiated agreements are included in the table below, not unilateral commitments.

Voluntary approaches	**Environmental policy field**	**Year introduced**
Sweden		
Research & development initiative on environmentally friendly vehicles	Air pollution; Climate change; Transport	1994
Pollution from shipping	Air pollution; Water pollution	1996
Environmental quality classes for lead in petrol	Air pollution	1994
Environmental quality classes for petrol	Air pollution; Climate change	1997
Energy efficiency agreements for individual firms	Energy efficiency	1994
Energy efficiency in industry	Energy efficiency	2004
Construction materials	Waste management	1995
Recycling of NiCd batteries	Waste management	1996
Recycling of office paper	Waste management	1998
Sludge, including use in agriculture	Waste management	1994

Chemical controls, including processes (various agreements)	Chemical control	1993
Pesticides	Chemical control	1995
Decontamination of closed petrol stations	Soil contamination	1997
Conservation of natural forests	Natural resources management	1998
UK		
Pesticides Safety Precautions Scheme	Health and safety	1957-86
'BASIS' (pesticides handling)	Health and safety	1978-
The Pesticides Voluntary initiative	Water pollution; Air pollution; Land contamination	2001
APEs (domestic fabric washing products)	Water pollution	1972-78
APEs (industrial washing products)	Water pollution	1995-
NTA (doemstic fabric washing products)	Water pollution	1975-
Recycled content of newsprint	Waste management	1991-
Farm Films Recovery Scheme (FFSR)	Waste management	1995-97
HFCs (aerosol industry)	Climate change	1996-
HFCs and PFCs (fire industry)	Climate change	1996-
HFCs (foams industry)	Climate change	1996-
Energy efficiency (chemical industry)	Climate change	1997-
Climate Change Agreements with 38 industrial associations	Climate change	2001

Sources: Jordan, Wurzel et al. (2003e); OECD (2003c); Naturvårdsverket (2000b); OECD (2004b, p. 51).

APPENDIX V

Municipal waste policy instruments at the national level in Sweden and England

Table V-A. Municipal waste policy instruments adopted at the national level in Sweden*

EPI type	Year of introduction and use 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
COMMAND-AND-CONTROL INSTRUMENTS											
Licenses/permits, including emission and process standards	Waste management site licensing (permitting procedure)			*Amended regulation (SFS 1998:899)*							
							Emissions standards landfill sites *(SFS 2001:512)*				
								Emissions standards waste incineration *(SFS 2002:1060)*			
Prohibition bans				Obligatory batteries collection *(SFS 1997:645)*							
								Ban on landfill of combustible waste *(SFS 2001:512 9 §) (combined with exemption licenses, NFS 2001:117)*			
											Ban on landfill of organic waste *(SFS 2001:512 10 §)*
Statutory targets	(None)										
ECONOMIC INSTRUMENTS											
Taxes and charges	Charge on imported aluminium cans (1983-2005) *(SFS 1983:847)*										
	Battery charge (1987-) (*SFS 1997:645*)										
						Landfill tax: EUR 27 per tonne *(SFS 1999:673)*		EUR 31 per tonne	EUR 40 per tonne		
Subsidies and grants				Local investment programmes (1998-2002) *(SFS 1998:23)*							
											Investment grant for sorting facilities *(SFS 2004:1387)*
Transferable permit systems	(None)										
Depost-refund systems	Deposit-refund system: cars (1975-) *(SFS 1975:348)*										
	Deposit-refund system: aluminium cans (1982-) *(SFS 1982:349)*									(Incorporated into producer responsibility for packaging in 2003)	
	Deposit-refund system: plastic bottles (1991-) *(SFS 1991:336)*									(Incorporated into producer responsibility for packaging in 2003)	
LIABILITY AND DAMAGE COMPENSATION											
Extended producer responsibility	Producer responsibility: packaging – recycling targets for 2001 (1994-) *(SFS 1994:1235)*		Amended; recycling targets for 2005 *(SFS 1997:185)*								
	Producer responsibility: newspaper – collection target for 2000 (1994-) *(SFS 1994:1205)*										

	Producer responsibility: car tyres – non-landfill disposal target (1994-) *(SFS 1994:1236)*										
				Producer responsibility: cars – recycling and reuse targets for 2006 and 2015 *(SFS 1997:788)*							
							Producer responsibility: electric and electronic equipment – no target *(SFS 2000:208)*				
EDUCATION AND INFORMATION											
Public campaigns							'Sopor.nu' – website with consumer information on recycling				
								'Hem till holken' – consumer information campaign on batteries collection			
Technology diffusion	(None)										
VOLUNTARY APPROACHES											
Unilateral commitments	(None)										
Negotiated agreements	(None)										
MANAGEMENT AND PLANNING											
Local plans	Statutory municipal waste regulations and waste plan (1991-) *(Env Code, 15 ch., 11 §)*										

*SFS numbers refer to the respective legislative codes (laws or ordinances), the NFS number refers to regulation adopted by Swedish EPA. The exchange rate used here is 9.31 SEK/EUR.
Sources: Environment Code; MoSD & SEPA websites; Gov. Bills 2002/03: 117 & 2004/05: 150; OECD (2004); SOU(2005).

Table V-B. Municipal waste policy instruments adopted at the national level in England*

EPI type	**Year of introduction and use** 1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
COMMAND-AND-CONTROL INSTRUMENTS											
Licenses/permits, including emission and process standards	Waste Management Licensing Regulations 1994 (1994-)										
								Landfill permits and emission standards (*Landfill Regulations 2002*)			
								Emission standards (*Waste Incineration Regulations 2002*)			
Prohibition bans	(None)										
Statutory targets							Statutory Performance Standards for the recycling of household waste (*Local Government (Best Value) Performance Indicators and Performance Standards Order 2001*)				
ECONOMIC INSTRUMENTS											
Taxes and charges		Landfill tax: EUR 4.80 per tonne non-inert waste; EUR 1.40 per tonne inert waste (*Landfill Tax Regulations 1996*)	EUR 4.80 per tonne non-inert waste; EUR 1.40 per tonne inert waste	EUR 4.80 per tonne non-inert waste; EUR 1.40 per tonne inert waste	EUR 6.90 per tonne non-inert waste; EUR 1.40 per tonne inert waste	EUR 7.59 per tonne non-inert waste; EUR 1.40 per tonne inert waste	EUR 8.28 per tonne non-inert waste; EUR 1.40 per tonne inert waste	EUR 8.97 per tonne non-inert waste; EUR 1.40 per tonne inert waste	EUR 9.66 per tonne non-inert waste; EUR 1.40 per tonne inert waste	EUR 10.35 per tonne non-inert waste; EUR 1.40 per tonne inert waste	Landfill tax: EUR 12.42 per tonne non-inert waste; EUR 1.40 per tonne inert waste

Subsidies and grants	Recycling Credits (1992-) (*Environmental Protection Act 1990*)										
		Landfill Tax Credit Scheme (*Landfill Tax Regulations 1996*)									
						Home Composting Scheme (WRAP)					
						Retailer Initiative and Innovation Fund (WRAP)					
						Materials Programme (WRAP): glass, Plastics, Paper, Wood, Organics, Aggregates					
						Recycling Fund (WRAP)					
						eQuip Residual Value Guarantee (WRAP)					
									National Waste Minimisation and Recycling Fund		
										Waste Research Programme (WIP)	
										Demonstrator Programme (WIP)	
										Community Sector Support Programme (WIP)	
											Waste Performance and Efficiency Grant
											Business Resource Efficiency and Waste Programme (BREW)
Transferable permit systems			Packaging Recovery Notes (implements packaging producer responsibility) (*Producer Responsibility Obligations Regulations 1997; Packaging Regulations 1998*)								
											Landfill Allowance Trading Scheme (*Waste and Emissions Trading Act 2003*)
Depost-refund systems	(None)										
LIABILITY AND DAMAGE COMPENSATION											
Extended producer responsibility									Producer responsibility: End-of-Life Vehicles producer responsibility (*End-of-Life Vehicles Regulations 2003*)		
EDUCATION AND INFORMATION											
Public campaigns	Rethink Rubbish										
						Recycle Now campaign (WRAP)					
						Home Composting Scheme (WRAP)					
						Real Nappy programme (WRAP)					
							Recycled Products Guide (WRAP)				
Technology diffusion						Materials Programme (WRAP): glass, Plastics, Paper, Wood, Organics, Aggregates					
						ROTATE (WRAP)					
									New Technologies: Supporter Programme (WIP)		
VOLUNTARY APPROACHES											
Unilateral commitments			Automotive Consortium on Recycling and Disposal								
Negotiated agreements						VA with Newspaper Publishers Association					
									VA with Direct Marketing Association		
MANAGEMENT AND PLANNING											
Local plans	Statutory Waste Recycling Plans										
	Statutory Waste Local Plan										
						Non-statutory municipal waste strategy					
											Statutory Joint Municipal Waste

											Management Strategy in two-tier areas (*Waste and Emissions Trading Act 2003*)

* The exchange rate used here is 0.69 GBP/EUR. 'WIP' refers to the instrument being part of the Waste Implementation Programme. 'WRAP' refers to the instrument being part of the Waste and Resources Action Programme.
Sources: DEFRA, DTI, EA, WRAP and LTCS websites; DETR (2000); Strategy Unit (2002); OECD (2002); Davoudi (2000).