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**THE IMPLEMENTATION OF EUROPEAN UNION
ENVIRONMENTAL POLICY: THE CASE OF THE PACKAGING
WASTE DIRECTIVE**

by

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A thesis submitted to the University of Plymouth
in partial fulfilment for the degree of

DOCTOR OF PHILOSOPHY

Department of Geographical Sciences
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September 2000

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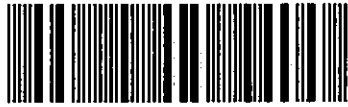
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ABSTRACT

THE IMPLEMENTATION OF EUROPEAN UNION ENVIRONMENTAL POLICY: THE CASE OF THE PACKAGING WASTE DIRECTIVE

This thesis provides a critical review of the processes shaping the implementation of European Union (EU) environmental policy. It focuses on two aspects of this dynamic, the interpretation of EU law by Member States and the use of legislative and price-based policy instruments to achieve policy objectives. The overall aim of the study is to examine the extent to which price-based regulation can contribute to the EU's policy objective of sustainable development. The focus for the research is the formulation of the Packaging and Packaging Waste Directive and its implementation in two Member States, Britain and Germany. A variety of research methods were employed, including literature and document searches, personal correspondences, telephone interviews, and postal surveys. The latter stage included a survey of British and German businesses affected by national packaging waste legislation.

The first major finding was that the methods used by Member States to implement EU requirements are a major determinant of the sustainability outcomes achieved. By adopting command-and-control legislation and punitive environmental charges, Germany has achieved high recycling rates and significant reductions in packaging consumption. Britain's market-led approach has struggled to achieve its environmental targets but has produced a relatively cost-efficient recycling system. However, the second major finding was that environmental charges have not altered industry behaviour significantly. Whilst German firms were found to be more actively involved in preventative waste management than their British counterparts, this has been brought about primarily by legislative provisions and the readiness of national authorities to resort to constrictive regulation. The main contribution of price-based regulation has instead been the generation of hypothecation revenue for pollution control. From these findings, a conceptual model outlining the sustainability outcomes produced by legislation and price-based regulation is developed and discussed.

From this evidence, it is concluded that the use of price-based regulation alongside state-determined implementation has led to some divergence in the sustainability outcomes achieved by EU environmental law. Moreover, the economic approach to environmental problems does little to resolve the fundamental conflicts of priorities between the EU's environmental agenda and its other policy domains. Some options for greater co-ordination of economic instruments at the EU level are suggested and evaluated. The thesis therefore provides a wide-ranging analysis of the practical application of price-based environmental regulation. Its primary contribution is that it assesses how political and practical issues combine to influence the implementation of environmental policy. Furthermore, by assessing EU policy in terms of its contribution to sustainable development, the study has sought to provide a holistic examination of the forces determining the success of the EU's environmental programme.

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LIST OF ABBREVIATIONS

ACP	Advisory Committee on Packaging
ARGUS	<i>Arbeitsgruppe Umweltstatistik</i>
BAT	Best Available Technology
BATNEEC	Best Available Technology Not Entailing Excessive Cost
BMU	<i>Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit</i> (German Environment Ministry)
CbvA	Cost-benefit versus Action
CEC	Commission of the European Communities
CEE	Central and Eastern Europe
COPAC	Consortium of the Packaging Chain
COREPER	Committee of Permanent Representatives
CORINE	Co-ordination of Information on the Environment
DETR	Department of the Environment, Transport and the Regions
DoE	Department of the Environment
DSD	Duales System Deutschland
DTI	Department of Trade and Industry
EAP	Environmental Action Programme
EC	European Community
ECJ	European Court of Justice
EEA	European Environment Agency
EEC	European Economic Community
EfW	Energy from Waste
EMAS	Environmental Management and Audit Scheme
EMU	Economic and Monetary Union
ENDS	Environmental Data Services
EP	European Parliament
ESC	Economic and Social Committee
ETR	Environmental Tax Reform
EU	European Union
EUROPEN	European Organisation for Packaging and the Environment
HCSCETRA	House of Commons Select Committee on Environment, Transport and Regional Affairs
IGC	Inter-Governmental Conference
IMPEL	Network for the Implementation and Enforcement of Community Law
INCPEN	Industry Council for Packaging and the Environment
IWM	Institute of Wastes Management
LA21	Local Agenda 21
LCA	Life Cycle Analysis (or Assessment)
LDC	Less Developed Country
LGMB	Local Government Management Board
LIFE	L'Instrument Financier pour l'Environment
MSB	Marginal Social Benefit
MSC	Marginal Social Cost
MRW	Materials Recycling Week
NIMBY	Not In My Back Yard
NIS	New Independent States (of the former Soviet Union)
NUDIST	Non-numerical Unstructured Data Indexing Sorting and Theorising
OECD	Organisation for Economic Co-operation and Development
OJEC	Official Journal of the European Communities
PPP	Polluter Pays Principle

PRG	Producer Responsibility Group
PRN	Packaging waste Recovery Note (also Producer Responsibility Note)
PWMS	Packaging Waste Management System
SEA	Single European Act
SEPA	Scottish Environmental Protection Agency
SME	Small-Medium Enterprise
SPSS	Statistical Package for the Social Sciences
TEU	Treaty on European Union
UK	United Kingdom
UN	United Nations
UNCED	United Nations Commission for Environment and Development
US(A)	United States of America
V-WRAG	VALPAK-Working Representatives Advisory Group
WCED	World Commission for Environment and Development
WEEE	Waste Electrical and Electronic Equipment
WMH	Waste Management Hierarchy
WMHA	Waste Management Hierarchy Action
WMHT	Waste Management Hierarchy Target

ACKNOWLEDGMENTS

This thesis would not have been possible or as enjoyable without the assistance of a great number of people. First and foremost, I would like to thank my Director of Studies, Professor Mark Blacksell, who has been unstinting in his support and in challenging me to greater achievements. I am also deeply grateful to Dr. Mark Wise for his encouragement, his meticulous eye for detail, and for his formidable appreciation of the labyrinths of EU politics. Moreover, they have both made researching and writing this thesis a genuine pleasure.

My sincerest thanks also go to the Institute of Wastes Management for their financial support for my fieldwork, and to the Environment Agency for providing information used in the surveys. However, nothing would have been possible without the patience and assistance of the hundreds of businesses and individuals that responded to my surveys and requests for information. Special mention here goes to the staff at the *Duales System Deutschland* for providing literature and answering my endless questions. This work has also benefited enormously from dialogue with other research institutions. I would particularly like to thank Juergen Gonser and Karin Jordan of the *Arbeitsgruppe Umweltstatistik* at the Technical University of Berlin; David, Denise, and Gill at Perchards; Markus Haverland from the University of Nijmegen; and Richard O'Doherty from the University of the West of England.

Back at home, I am extremely grateful to my fellow postgrads for their advice, criticism and support. As my academic siblings and thesis proof readers, Jon Shaw and Bettina van Hoven-Iganski deserve a special mention, as does Karl Martin Born for his impeccable translations and insights into all things German. Thanks also to other members of the postgraduate community; Erik Meyles, Francien van Soest, Roger Levermore, Catherine Reid, Jon Vernon, Astrid Blum, Liz Hunt, Amy Burgess, Tony Massey and Emma Vickery. The Department's technical departments have also been enormously helpful throughout this study; special thanks here go to Tim, Brian and Jamie in the Cartographic Resources Unit, also Pauline, Adrian and David in IT.

Finally, a personal thank you to my wife, Rebecca, for being so patient, forgiving and supportive, and for enduring academic loneliness with great fortitude. This thesis is dedicated to the memory of my mother and to my father.

AUTHOR'S DECLARATION

At no time during the registration for the degree of Doctor of Philosophy has the author been registered for any other University award.

The study was financed with the aid of a studentship from the Higher Education Funding Council for England.

A programme of advanced study was undertaken which included a Master's degree in Social Research.

Relevant seminars and conferences were attended at which work was regularly presented; external institutions were contacted for consultation purposes and a series of papers were prepared for publication.

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Paperpack Ltd.

Perchards Packaging Consultancy

RIGK MBH

Robin Teverson, MEP and Ken Collins, MEP

The British Retail Consortium

The European Organisation for Packaging and the Environment (EUROPEN)

The Institute of Wastes Management

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VALPAK

Signed 

Date 15. 9. 2000

Chapter One

Introduction

1.1 Issues in European Union Environmental Policy

For many years the European Union (EU) Member States generally held the view that environmental protection was a national concern which did not warrant more than occasional supra-national action. The Treaty of Rome creating the European Economic Community did not originally prioritise or even include a reference to environmental concerns (Zito, 2000). It was not until the First Environmental Action Programme (EAP) in 1973 that the need for co-ordinated action to combat trans-boundary pollution and distortions of the Single Market caused by national environmental standards was formally recognised. Moreover, it was only with the Single European Act (SEA) in 1986 that environmental protection was officially incorporated within the EU remit (Blacksell, 1994). Considering the late arrival of environmental policy on the international political stage and its potential conflicts with well-established economic and social priorities, it is not surprising that EU decision-makers have struggled to craft an effective and coherent body of environmental policy (Bailey, 1999a). These challenges have spanned not only the technicalities of defining policy aims and the 'best' methods of achieving them - issues common to all policy programmes (Segerson, 1996; O'Riordan and Voisey, 1998) - but also their assimilation into the unique and complex political structure of the EU (Howe, 1996; Haigh, 1998).

Whilst the EU has made increasingly clear commitments to sustainable development as a grand policy goal (Commission of the European Communities (CEC), 1992a), agreeing specific courses of action involves often complex negotiations between the EU's constituent governments, its permanent institutions and other significant stakeholders. The situation is further complicated by the manner in which EU law is put into action, in that control over practical implementation is vested almost entirely with the Member States. As such, 'EU policy only comes to life in the member states and thus only has significance to the extent that it goads or galvanises national institutions, organisations and citizens to act' (Lowe and Ward, 1998a: 4). A by-product of this state-led approach, however, has been that the way EU policies are

implemented often varies widely between Member States despite their acceptance of ostensibly common legislative standards.

Although such issues have long been part and parcel of policy-making in the EU, they have taken on new relevance with the Union's increasing predisposition towards price-based environmental regulation (CEC, 1992a). Price-based regulation is specifically designed to integrate the full social costs of environmental exploitation into the economies of the Member States. Although they remain bound by the terms of the Treaty, the signals sent to national markets using price-based regulation are determined primarily by state or regional authorities. Where Member States hold divergent views on both the prioritisation and conduct of environmental policy, any significant move towards price-based regulation may further embed these differences (Haverland, 1999). Moreover, because price-based regulation is a comparatively 'new' approach to environmental policy, its practical efficacy remains relatively untested. The first aim of this thesis, therefore, is to explore the contribution made by price-based policy instruments to the success of the EU environmental programme. The second aim is to investigate their operation within the EU's distinctive policy formulation and implementation procedures. In order to examine these issues, the study analyses the Packaging and Packaging Waste Directive (94/62/EC), an EU initiative which most Member States have implemented using price-based regulatory techniques. Before embarking on this analysis, however, this chapter outlines the direction the thesis will take. It develops the key conceptual themes under investigation, introduces the legislation being studied, and sets out the objectives and structure of the thesis.

1.2 Themes of the Study: Policy Instruments and Political Influences

1.2.1 Policy Instruments: Price-based Environmental Regulation¹

There has been longstanding academic interest in environmental policy instruments, the general aim of which has been to understand their environmental and economic implications. One of the key discussions has been the relative merits of legislative

¹ Several variants of the term 'price-based regulation' are used throughout this thesis, including economic instruments and environmental charges. Although these should be taken as having an equivalent meaning, other variants, notably market-based regulation and environmental taxes are used to refer to specific features of price-based regulation, for example, whether pollution charges are determined primarily by market forces or government intervention.

standards and economic instruments². The main attribute of the legislative approach is that it imposes clear and enforceable standards and responsibilities. By contrast, price-based regulation seeks to ascribe monetary values to environmental resources in order to encourage their prudent utilisation (under the Polluter Pays Principle, or PPP). There has been particular interest in economic instruments recently, first, because of the perceived failure of legislative standards to alleviate environmental degradation (Pearce *et al.*, 1989; Pearce and Turner, 1990; CEC, 1992a) and, second, as a consequence of the suggestion that economic instruments can achieve environmental solutions in a cost-effective manner (Baumol and Oates, 1988; Tietenberg, 1990; Bahn, 1999). This belief is supported by a large, predominantly theoretical, literature which suggests that price-based regulation will encourage industries to reduce the scale of their environmentally-damaging activities³.

Despite the theoretical advantages of price-based policies, the economic approach to environmental issues has been adopted in surprisingly few policy programmes (Tietenberg, 1990; Jacobs, 1991; Huppel *et al.*, 1992; Howe, 1996). Although this can be partly explained by political concerns over their inflationary impact and environmental efficiency (Helm, 1998), the corollary is that there has been comparatively little empirical research examining the practical effects of price-based regulation. Instead, most studies have either focused on *modelling* the impact of economic instruments (for example, Pearce *et al.*, 1989; 1993; Brisson, 1993; van den Bergh, 1996; Bohm, 1997; Ekins, 1997; Xepapadeas and de Zeeuw, 1999) or policy issues (for example, Levenson, 1993; Fenton and Hanley, 1995; Defeuilley and Godard, 1997; Powell and Craighill, 1997; Sinclair and Fenton, 1997; Porter, 1998, Turner *et al.*, 1998)⁴. Only a few studies, notably Labatt (1991; 1997a; 1997b), have

² Authors discussing legislative regulation include Collins and Earnshaw (1993), Lévêque (1995; 1996a), Skea (1995) and Whiston and Glachant (1996). Those examining price-based regulation include Baumol and Oates (1979; 1988), Pearce *et al.*, (1989; 1993), Hahn (1989), Tietenberg (1990), Pearce and Turner (1992; 1993), Helm (1993; 1998), Turner (1993), Turner and Pearce (1993), Goddard (1995) and Turner *et al.* (1998). Notable critiques of price-based regulation include Daly and Cobb (1990), Daly (1992), Beder (1996) and More *et al.* (1996).

³ For example, Pigou (1920), Baumol and Oates (1979; 1988), Schelling (1984), Pearce *et al.* (1989; 1993), Hahn (1989), Turner and Pearce (1990; 1993), Huppel *et al.* (1992), Repetto *et al.* (1992), Stavins and Whitehead (1992), Brisson (1993), Goddard (1995) and Gersbach and Glazer (1999).

⁴ However, a number of studies have debated the efficacy of environmental charges as a means of changing business behaviour on the basis of either logical reasoning or policy analysis. Notable works include Opschoor and Vos (1988), Hahn (1989), Jacobs (1991), Goddard (1995), Pearce *et al.* (1993) and Beder (1996). Several reports conducted for the European Commission and national governments also assess the impact of price-based policies in this manner, for example, the Department of Trade and Industry (DTI)/Department of the Environment (DoE) (1991; 1992), DoE (1993), CEC (1994) and Organisation of Economic Co-operation and Development (OECD) (1994).

specifically analysed business responses to PPP-based charges and referenced these against defined environmental targets. With several influential governments now beginning to experiment extensively with price-based environmental regulation (Ekins, 1993; Segerson, 1996; Vogel, 1997)⁵, this lack of empirical verification is being exposed as a serious deficit in the literature. The first original contribution of this thesis, therefore, is its evaluation of the practical benefits of price-based environmental regulation.

1.2.2 Formulating and Implementing Environmental Policy in the EU

The study's second theme concerns the political dynamics of environmental policy-making in the EU. Böhmer-Christiansen (1994), Demmke (1994) and Haas (1999) argue that understanding the implementation of *any* policy is impossible without first appreciating the circumstances in which it is formulated, as political issues inevitably influence the direction policies take, the scale of their ambitions and the manner in which they are implemented. Although sustainable development has become almost a universal paradigm, its expansive guidelines have proven susceptible to politically and economically motivated reinterpretation. Thus, it has proven difficult to translate sustainable development's conceptualisation of enduring pathways for human society into clear courses of action. The literature discussing sustainable development is immense and, out of necessity, this thesis considers the debate only briefly⁶. Moreover, if environmental problems are to be addressed convincingly, the debate must ultimately be founded on empirical evidence rather than abstract theorising (Ekins, 1993). That said, the thesis seeks to recognise the complexity of the sustainability debate and its vulnerability to political manipulation.

The EU environmental programme is also profoundly influenced by its complex political agenda. Although the Union aspires to far-reaching economic and policy integration, it remains a grouping of often fiercely independent states (Wise and Gibb, 1993). The tensions between the EU's expansive integration agenda and its desire to defend the sovereignty of its Member States makes the development of common

⁵ Within the EU context alone, the Waste Electrical and Electronic Equipment (WEEE) and End-of-life Vehicles Directives are both likely to be implemented using economic instruments (Materials Recycling Week (MRW), 1999a; 1999b).

⁶ Pezzoli (1997) provides the most comprehensive review of the sustainable development literature. Other important contributions include Ekins (1993), Turner (1993), Redclift (1987; 1992), Pearce *et al.* (1989; 1993) and O'Riordan and Voisey (1998).

environmental policies a highly intricate process. Although this applies to all aspects of EU activity, the environment is recognised as a particularly difficult area to coordinate because of the complex nature of trans-national pollution, the extent that environmental initiatives permeate traditionally national policy domains, and because of conflicts between the EU's environmental and economic priorities (Collins and Earnshaw, 1993, Weale, 1996).

The politics of EU environmental policy centres on three areas of potential conflict; (i) decisions on the acceptance of EU action, (ii) the political dynamics of negotiating environmental laws, and (iii) the process of practical implementation in the Member States. Prior to the SEA, the EU was only legally entitled to enact environmental policies in order to protect free trade in the Common Market. This made it all the more remarkable that a substantial body of environmental legislation was developed during this period (Lowe and Ward, 1998a). The Maastricht and Amsterdam Treaties have further augmented the powers conferred by the SEA such that environmental protection is now officially 'an essential objective' of the EU⁷. However, EU involvement can still only be justified where environmental objectives or the defence of the Single Market can be more effectively achieved through EU rather than state action (Toth, 1994). The management of this issue, by means of the hotly debated subsidiarity principle, is examined in Chapter three.

The issue of policy formulation is complicated by the fact that the EU is not a unitary state. It might be more accurately defined as a process whose product is the aggregated and transformed ideas of its constituent national powers (Zito, 2000). This means that the negotiation of policies is often a very state-centred and interest-led process. The main tension is therefore between the desire to achieve unanimity on key issues and the ambition to have national agendas elevated at the EU arena. Again this is not unique to environmental policy but is accentuated by the presence of defined environmental 'leader' and 'laggard' Member states in the Council. In terms of policy outcome, the tension is between 'lowest-common-denominator' bargaining and a more 'entrepreneurial' style of decision-making seeking to promote greater integration and high environmental standards (Collins and Earnshaw, 1993; Sbragia, 1996; Zito, 2000).

⁷ As noted by the European Court of Justice in the Danish Bottles Case (302/86, ECR 4607) in 1988.

The final point of contention, policy implementation, arises because the majority of environmental policy is legally enacted in the form of directives. This means that Member States are bound in terms of the overall objectives to be achieved but retain the right to determine the detailed arrangements for putting them into practice (Jordan, 1999). Whilst this deliberately makes EU law flexible without allowing national authorities to disavow it entirely, it has led to frequent disputes on the precise timing and extent of implementation by Member States. Moreover, the success of directives is measured almost entirely in terms of legislative standards, a relatively blunt method for assessing *how* Member States achieve EU standards and, therefore, the overall contribution of national policies to sustainable development.

These factors have combined to produce complex and lengthy policy-implementation procedures, and often severe dislocations between the aims enunciated by the EAPs and the practical results achieved (Collins and Earnshaw, 1993; Krämer, 1996; EUR-OP News, 2000). However, whilst some commentators criticise the EU for being an incomplete polity and for its excessive flexibility in relation to policy implementation (W. Wallace, 1996; Krämer, 1996), others stress that the level of integration achieved within the EU is an impressive political achievement (Wise and Gibb, 1993; Scott *et al.*, 1994). The thesis' second original contribution is its examination of how the EU's political structure affects the efficacy of price-based environmental regulation. It therefore assesses how technical and political factors inter-twine to determine environmental policy outcome. Whilst previous studies have also examined aspects of this issue (for example, Demmke, 1997; O'Riordan, 1997; Lowe and Ward, 1998a; Turner *et al.*, 1998), most, somewhat artificially, have treated politics and policy instruments as separate issues.

1.3 Background to the Packaging Waste Directive

The Packaging and Packaging Waste Directive was formally adopted by the Council of Ministers and European Parliament in December 1994 (Official Journal of the European Communities (OJEC), 1994). Its primary aim was to harmonise EU recycling laws on packaging waste following the introduction of the German *Verpackungsverordnung* (Packaging Ordinance) in 1991 (London and Llamas, 1994; Waite, 1995). Fearing that the recycling targets and costs imposed by the Ordinance would become technical obstacles to the free trade of packaged goods in the Single

Market, other states and the Commission pressed for harmonising legislation under Article 100a of the EU Treaty⁸. The Packaging Directive requires all Member States to introduce measures:

aimed, as a first priority, at preventing the production of packaging waste and, as additional fundamental principles, at reusing packaging, at recycling and other forms of recovering packaging waste and, hence, at reducing the final disposal of such waste (Article 1) (OJEC, 1994: 12)⁹.

However, although the Directive seeks to promote a range of waste management objectives, the only commitments actually quantified were those relating to the recovery and recycling of packaging waste (50-65% and 25-45% respectively). Following heated negotiations in the Council of Ministers and European Parliament (EP), the prevention and re-use of packaging waste and the development end-use markets for recycle were only included as general conditions in the final Directive's 'Essential Requirements' (Annex II) and Article 6. Golub (1996) cites this as evidence of lowest-common-denominator bargaining within EU environmental policy.

Two points are immediately obvious from this. First, in common with much EU legislation, the Directive sought legislative approximation rather than total harmonisation. To further this aim, banded targets and derogations were employed to cater for the specific exigencies and capabilities of individual Member States (Bailey, 1999a). In addition, the Directive itself was framed primarily in the form of legislative standards rather than price-based regulation. However, Article 15 provides the guidance that: 'acting on the basis of the relevant provisions of the Treaty, the Council adopts economic instruments to promote the implementation of the objectives set by this Directive' (OJEC, 1994: 16). It therefore becomes apparent that many Member States anticipated implementing the Directive's requirements using some form of price-based regulation. Because of this and the practical complexities of waste recycling, the Packaging Directive has, slightly inadvertently, become a prominent example of price-based regulation in the EU, making it an ideal focus for investigating the impact of

⁸ This thesis refers to Articles 100 and 100a although they were amended to Articles 94 and 95 in the Amsterdam Treaty, as the former were still in force at the time the Packaging Directive was negotiated.

⁹ Under the Directive, recycling is defined as 'the reprocessing in a production process of the waste materials for the original purpose or for other purposes including organic recycling but excluding energy recovery'. Recovery is defined as 'the use of packaging waste as a means to generate energy through direct incineration with or without other waste but with recovery of the heat' (OJEC, 1994: 13).

economic instruments on business behaviour and their compatibility with the EU's state-led approach to policy implementation.

1.4 Aims and Objectives of the Study

Against this background, the central objective of this thesis is to use evidence from the Packaging Waste Directive to help contribute towards a more general understanding of the extent to which price-based policy instruments promote sustainable development within the EU. Recognising that EU environmental policies are influenced by a range of technical and political factors, the study analyses how, in practice, interactions between the two determine policy outcome. It therefore critically evaluates the negotiation, transposition and implementation of the Packaging Directive as well as the translation of price-based policies from theory into practice. The research has three key objectives:

- To assess, within the example of the Packaging Directive, the efficacy of the EU's state-led style of policy negotiation and implementation in relation to the promotion of sustainable development;
- To investigate the extent to which price-based policy instruments have encouraged sustainable business practices;
- To assess the compatibility of the EU's political and decision-making structures with the effective operation of price-based environmental regulation.

1.5 Structure of the Thesis

To examine these issues, the thesis is structured into eight further chapters. Chapter two examines the general theoretical context of environmental policy-making by investigating the general aims of environmental policy and the implementing mechanisms used to achieve them. The chapter begins by outlining the genesis of modern environmental thinking and its evolution from the limits-to-growth hypothesis to sustainable development. This is followed by a critical review of sustainable development and its acceptance as the main framework for contemporary environmental programmes (Redclift, 1987; 1992; Pezzoli, 1997; Turner, 1997). The selection of policy instruments to promote sustainable development is then investigated, paying particular attention to the conceptual merits of legislative standards

and price-based instruments. Chapter three then discusses the politics of environmental policy in the EU. It begins by reviewing the justifications behind the EU environmental programme and its evolution since the First EAP. The nature of European integration and the forces shaping the creation and implementation of environmental policies are then examined. This is followed by a discussion of the practical stages of policy negotiation and enforcement. Throughout the review, the tensions between the EU's economic and environmental priorities and between its main decision-making bodies are highlighted along with their implications for environmental policy outcome.

Before the thesis considers the expression of these tensions within the Packaging Directive, Chapter four describes the methods used to collect and analyse the research's primary and secondary data. This section emphasises the compilation of secondary material as a framework for understanding the general policy context, the use of qualitative data to formulate research hypotheses and, finally, the utilisation of quantitative techniques to test their validity. The secondary data used in the study were obtained from a variety of official documents, including government consultations, policy papers, industry submissions and academic analyses. The main primary data were derived from two surveys, the first conducted with reprocessing businesses in the UK and the second with 1800 businesses affected by national packaging legislation.

The remainder of the thesis examines the use of economic instruments to implement the Packaging Waste Directive. Chapter five provides a general overview of the Directive's negotiation and transposition, then compares the implementation methodologies employed in two Member States, Britain and Germany. It considers the extent to which policy implementation in each country has been shaped by prevailing institutional structures and political stances (see also Haigh and Lanigan, 1995; Haigh, 1998; Lowe and Ward, 1998a), then argues that two distinctive and ideologically-driven models of packaging waste management have emerged. The UK model, it contends, reflects the British government's desire to achieve EU standards in a cost-effective manner using market-based regulation. By contrast, the German approach has been characterised by stringent environmental standards and the instigation of 'command-and-control' legislation (Haverland, 1999). It is further argued that these policy styles are evident in the way each country has applied price-based environmental regulation. The chapter concludes by assessing each system in terms of its ability to

produce environmentally and economically efficient systems of packaging waste management.

Chapters six and seven present the primary data derived from the survey of UK and German businesses and examine how the two regulatory models have influenced the environmental behaviour of industries affected by the Directive. Chapter six sets the scene by contrasting the packaging waste management practices adopted by British and German businesses. Chapter seven then assesses the extent to which economic instruments have influenced business actions in each country. As a prelude to this, the literature examining the merits of price-based environmental regulation is revisited, then its assertions are tested using data from the study's main survey. On the basis of these results, the nature of corporate responses to legislative and price-based policy instruments and the contribution of each to sustainable development are reflected on.

Chapter eight returns to the policy-making dimension of the study to analyse the compatibility of economic instruments with the decision-making structures of the EU. It investigates the degree to which the Packaging Waste Directive has led to the convergence or divergence of national policies, then explores whether price-based regulation has improved the implementation of EU policies. Following this, the chapter discusses the incorporation of the EU's sustainability ambitions within a supranational system primarily geared towards economic and trade development. Finally, Chapter nine summarises the study's findings and reviews its limitations. In so doing, it examines future challenges for price-based regulation in the EU and highlights areas of potential future research.

Chapter Two

Aims and Mechanisms of Environmental Policy

2.1 Introduction

Chapter one identified the main aim of this thesis as an evaluation of the success of European Union environmental policy in promoting sustainable development and, in particular, the contribution of price-based regulation to this objective. Before the analysis proceeds, however, the purpose of this chapter is to establish the conceptual basis of the study. Policy analysts usually identify two principal components in any policy-making process, the determination of desired objectives and the selection of policy mechanisms to secure these goals (Segerson, 1996; Auty and Tribe, 1997). Cognitively, this may appear straightforward; without aims, policy intervention will be largely directionless, without effective implementing mechanisms, even the most consistent principles can have little substance. However, as the subsequent analysis demonstrates, the interpretation of these seemingly self-evident maxims is keenly contested within the polemics of environmental policy.

The main purpose of this chapter, therefore, is to review the debate on these two issues. The first section explores the literature discussing the emergence of sustainable development as the principal aim of international and EU environmental policy (see also Pezzoli, 1997). Various interpretations of sustainable development are discussed and its acceptance as the predominant paradigm of modern environmental management is analysed. Whilst it is recognised that sustainable development is an immensely fluid term, an appreciation of its various interpretations is central to understanding the basis of EU environmental policy and, moreover, the relationship between ecological protection and economic development. The second half of the chapter investigates the selection and application of environmental policy instruments. After reviewing the main policy mechanisms available, the use of price-based instruments within environmental policy is examined. Although environmental economists have advocated price-based environmental regulation for nearly twenty years, uncertainties over its environmental benefits and economic and political impact have meant that the technique has only recently gained real policy currency. Nonetheless, the indications

are that the EU sees price-based regulation as critical to the future of its environmental programme. The remainder of the chapter therefore examines the theoretical arguments for and against price- and market-based instruments as a method for integrating economic and environmental policy (Pearce *et al.*, 1989; Gibbs and Healey, 1997), along with the reasons behind their belated acceptance by environmental policy makers.

2.2 Policy Aims and Objectives

2.2.1 The Early Years of the Modern Environmental Movement

Though the roots of the environmental movement can be traced back many centuries (O’Riordan and Voisey, 1998; Pepper, 1986), its modern manifestation has been profoundly shaped by three works; Carson’s *Silent Spring* (Carson, 1962), Hardin’s *The Tragedy of the Commons* (Hardin, 1968), and the Club of Rome’s *Limits to Growth* report (Meadows *et al.*, 1972). Carson’s work, examining commercial agriculture, led her to conclude that existing farming practices were attempting to control nature in a technocentric manner - in order to maximise short-term productivity - without nurturing an accompanying appreciation of the environment’s resilience and assimilative capacity (Carson, 1962). In order to prevent major damage to the environment, she argued, industrial and agricultural processes needed to become less reliant on technological progress and more sensitive to ecological factors. Garrett Hardin’s work built upon these ideas and reinforced two further issues central to environmentalist thinking, the notion of carrying capacity and the self-destructive tendencies of individuals operating for private gain (Hardin, 1968). As his metaphor for the global environment, Hardin envisaged a piece of common land, which, he argued, could support a certain number of animals without losing its productive capacity. In other words, it had a finite carrying capacity. If one herdsman increased the size of his herd in order to enhance his personal yield and, by so doing, exceeded the commons’ overall carrying capacity for livestock, Hardin argued that degradation and loss of productive capacity would set in. However, the herdsman would benefit exclusively from having more animals but the overall loss to the commons would be shared amongst all parties, leaving the individual better off. Furthermore, on the basis of individual gain, the logical course of action is for each herdsman repeatedly to make the same choice, thereby moving the commons ecosystem from a position of stability towards one of rapid collapse. By realigning his analogy back to global issues, Hardin concluded that strong government policy rather

than reliance on individual actions was essential to protect the long-term sustainability of the global commons.

However, it was The Club of Rome Report that brought such concepts to widespread political and public attention. The report argued that existing planning systems were fundamentally geared towards understanding individual components of the global system - as determined by economic need - rather than their interactions, and therefore constituted neither a holistic nor a realistic consideration of human stresses on the natural environment (also Odum, 1970; Dahl, 1996). The *Limits* model predicted that widespread ecological and economic collapse would occur within 100 years if existing patterns of population growth, industrial production, resource exploitation and pollution continued unchecked. The report's fundamental notion, that technocentric industrial processes and spiralling population would combine to exceed the earth's carrying capacity, was reinforced by other neo-Malthusian visions of society's breakdown (for example, Ehrlich, 1971). Most subscribed to the view that the predicted disaster could only be averted by rejecting the previously uncontested orthodoxy of unending economic expansion and through strict population control measures (Schumacher, 1973; Pezzoli, 1997).

Whilst the concepts, methodology and predictions of the 'zero-growth' proponents were criticised by commentators who saw technological advancement as able to surmount any limits to growth (for example, Beckerman, 1974; 1995)¹, the *Limits* conclusions sparked a fierce academic debate and inspired the world's politicians publicly to re-appraise the relationship between growth and environmental protection at the United Nations Conference on the Human Environment in Stockholm in 1972 (O'Riordan, 1976). Pezzoli (1997) argues, however, that the focus throughout the 1970s was primarily on the reactive cleansing of the mess caused by unfettered industrialisation rather than the wholesale fusion of economic and environmental policies (see Chapter 3 for a discussion of the EU approach). Whilst the environment has never again been entirely removed from the political agenda², the global recession of the 1970s and early

¹ A special issue of *Futures* (Science Policy Research Unit, 1973) also presented an extensive critique of the methodologies employed by the *Limits* model. See also Weintraub *et al.* (1974).

² For example, the EU EAPs produced a large body of pollution-control legislation during the 1970s even though environmental policy had no basis in the EU Treaty until 1986 (see Chapter 3). Redclift (1987) also cites the World Conservation Strategy and the South Pacific Commission, both established in 1980.

1980s forced governments to concentrate principally upon their economic woes. It was not until the mid-1980s that environmental issues again captured the political limelight.

2.2.2 *The Brundtland Report and Sustainable Development*

In 1987 the World Commission on Environment and Development (WCED) published the report *Our Common Future* (the Brundtland Report). As with its predecessor, the *Limits* Report, but with firmer evidence of such phenomena as global warming and ozone layer depletion, it highlighted the need for humanity to reconsider its relationship with the natural environment. However, rather than returning to a neo-Malthusian population control philosophy, it attempted to blend economic development and environmental protection under the banner of *sustainable development* (WCED, 1987). This it defined as:

Development that meets the needs of the present without compromising the ability of future generations to meet their own needs (quoted in Haigh, 1998: 69).

Brundtland's interpretation of sustainable development contained four major themes; carrying capacity, quality of life, inter- and intra-generational equity, and consultation and partnership (Bennett and Patel, 1995; Clark *et al.*, 1993).

Whilst carrying capacity is an obvious extension of Hardin's work and the *Limits* Report, numerous commentators have attempted to clarify the practical implications of the concept (Pearce *et al.*, 1989; 1993; Daly and Cobb, 1990; Jacobs and Stott, 1992). As their basis for so doing, they defined ecosystem behaviour in terms of the laws of thermodynamics. The first states that matter and energy can neither be created nor destroyed but only transformed. The second states that whenever work is done, the amount of useable energy declines until entropy occurs. Within the global system, the only significant energy input is from the Sun - in economic terms, this is the Earth's *income* - while the natural resources available can be equated to its *capital* (Pearce *et al.*, 1989). Daly and Cobb (1990) argue that natural processes are low entropy, in that they take mainly solar-based energy and are net contributors to natural capital, but that production processes extract high levels of natural resources for the accumulation of human capital and, hence, are high entropy. They argue that a sustainable economy should live principally from the earth's income rather than by depleting its capital stock.

Furthermore, because natural capital is not an homogenous item but is instead comprised of innumerable, inter-linked and often non-renewable resources, a sustainable system needs to anticipate environmental problems rather than react to them (Pearce *et al.*, 1989). Implicit within the concept of carrying capacity, therefore, is the notion that economies and societies must be sustainable over an extended period of time. This is termed the concept of futurity (WCED, 1987). Whilst substantial disagreements remain as to what constitutes a sustainable level of natural capital and the desired direction of future human development (discussed in later sections), technocentric models which deny the very possibility of carrying capacities rarely have much credence in modern sustainability thinking (Ekins, 1993). As such, sustainable development argues that economic expansion needs to be placed within its ecological context rather than being considered as independent of it. As Dahl (1996: 85) notes:

The narrow criteria of financial profitability for corporations give ... the appearance of success, whilst permitting many costs to be externalised to the whole community, leaving society itself deeply in debt ... The economic system may seem rational and internally consistent, but it does not reflect reality.

The Brundtland Report's second major theme was that society should be judged against quality-of-life criteria that are broader than those relating to economic welfare within the formal economy (Redclift, 1987). Quality-of-life measures, Pearce *et al.* (1993) suggest, should include the advance of personal freedoms, self-esteem and self-respect (also Daly, 1992). Sustainable development therefore attempts to ingrain a greater sense of society, rather than just economy, within future policies. Closely allied to this is the notion of equity, which Pearce *et al.* (1989) argue encompasses both the present generation (intra-generational equity) and the fair treatment of future generations (inter-generational equity). Inter-generational equity is very much a distributional re-iteration of Brundtland's statement that the present generation should leave at least as much capital (human and natural) as it inherited for future generations. In terms of intra-generational equity, many authors see sustainability as impossible to achieve whilst enormous wealth disparities exist between the developed Northern countries and the less developed South and, moreover, while global economic systems reinforce these imbalances (Redclift, 1987; 1996; Pearce *et al.*, 1990; Jäger *et al.*, 1995). The strong redistributive element of sustainable development therefore differs fundamentally from

the emphasis on wealth creation and its natural diffusion through free market forces implicit in neo-classical models of economic growth.

Finally, Brundtland stressed the importance of consultation and partnership to sustainable development (Myers and Macnaghten, 1998). The report argued that neither market forces nor regulation could achieve the sustainability transition without widespread democratisation and public support for the policy commitments that accompany sustainable development (Bennett and Patel, 1995, Agyeman and Evans, 1997). Many therefore see locally focused policies rather than top-down dictates as the key to constructing sustainable development (Porritt, 1994; Gibbs *et al.*, 1998; Gerelli, 1995). However, whilst supporting this idea for some sustainability problems (Dovers 1997), others view the lack of power, finance and knowledge available within local authorities, along with their tendency to operate for local rather than global benefit, as major impediments to a locally-fostered sustainability strategy (Gibbs *et al.*, 1998; Voisey *et al.*, 1996; Burgess *et al.*, 1998; Marvin and Guy, 1997). They therefore point out that a strong institutional and integrative framework is needed from higher tiers of government in order to support local efforts.

So whilst Brundtland's vision of sustainable development defined a more reflective view of humanity's relations with the natural environment, it neither aspired to, nor presented, a comprehensive practical plan. Lee (1993), for example, sees sustainable development more as an appropriate vision of human endeavour than a policy programme, whilst Jäger *et al.* (1995: 14) view it simply as a basis for 'fostering ... "sustainable" synergies between environmental, economic and social policies.' What is clear, however, is that there is a distinction between the terms sustainability and sustainable development. Sustainability, Jacobs and Stott (1992) assert, focuses almost entirely on ecological concerns and minimum acceptable levels of environmental quality within an implicitly constrained economy, whilst sustainable development emphasises a new form of development de-coupled from increased resource consumption and pollution. As such, human welfare is a more prominent consideration within sustainable development. For some, economic growth is even a vital component of sustainable development's goal of social redistribution (Jongma, 1995). The complex relationship between economic growth and sustainable development is discussed further in the following section.

Though Brundtland's articulation of sustainable development is couched more in the terminology of vision than practical action, it nonetheless provided national governments and the international community with a basis for operationalising the linkages between economy and environment (Jäger *et al.*, 1995). However, this in itself presents practical difficulties. First, because many environmental problems are inherently international in character, respected trans-national institutions are required to co-ordinate the actions of national governments towards a 'common' goal of sustainability. Gerelli (1995) therefore argues that well-organised supra-national groupings such as the EU have a significant role to play in defining strategies for sustainable development. Second, practical policies are needed to promote Brundtland's principles. This issue is the subject of intense debate and is reviewed in the second half of the chapter. Nevertheless, several commentators have attempted to define in broad terms how sustainable development might be achieved (see, for example O'Riordan and Jäger 1995)³. Furthermore, sustainable development takes an ambiguous stance on the extent to which economic growth is a desirable policy goal and, therefore, remains vulnerable to dispute and reconstitution by nation states seeking to forward their national ambitions. Finally, despite the pressing need to communicate progress towards sustainable development (Cheatle, 1995; Brugman, 1997; Sterling, 1996), it has proven difficult to find accurate, meaningful and objective measures⁴.

Therefore, despite the immense contribution of the Brundtland report in bringing environmental issues to the global political arena, attempts to translate its principles into practical courses of action have been beset by numerous difficulties. Moreover, the sustainable development debate has extended beyond technical details into fundamental discussions on the direction sustainable policies should take. The next section reviews the varying perspectives on sustainable development contained within the literature.

³ These include directional spending arising out of ecotaxation to encourage, for example, job creation and health care, the creation of Local Agenda 21 networks to disseminate 'best' sustainability practices, and the decentralisation of economic activity.

⁴ See, for example the indicators produced by the UK Department of the Environment (DoE) *Indicators of Sustainable Development for the United Kingdom*, (DoE, 1996a). Whilst environmental indicators are an integral part of the report, it continues to support policies aiming towards economic expansion.

2.2.3 Perspectives on Sustainable Development

Although Brundtland's version of sustainable development remains the most widely quoted, over 70 definitions of the concept can be found (Pearce *et al.*, 1989; Kirkby *et al.*, 1995). In many ways, these disagreements are almost inevitable, since it is impossible to decipher such concepts as growth, development, equity and futurity without becoming embroiled in complex subjective and ethical judgements (Pearce *et al.*, 1989; Redclift, 1992). Fundamentally, the debate divides between those who support incremental changes to existing methods for valuing environmental resources (the economic view) and those who seek the wholesale ethical and institutional restructuring of society (the ecological and equity views)⁵. Reflecting this diversity, Turner (1993) and Gibbs *et al.* (1998) portray the debate as a spectrum between weak and strong sustainability using the typology shown in Table 2.1. However, though broad similarities exist between the economic view and weak sustainability, it would be inaccurate to classify them as entirely analogous because sustainable development embraces such a wide range of ambitions.

The economic view is best encapsulated by the work of Pearce *et al.* (1989; 1990; 1993), who present sustainable development as a positive means of managing growth. They argue that although economic production and consumption must be sustainable, zero growth is not a credible alternative because ecological processes cannot always take precedence over human welfare (also Castro, 1972). Thus economic expansion is important for achieving the equity objectives of sustainable development if it leads to improved material standards of living for the world's poor (Barbier, 1987). They nonetheless stress the importance of maintaining critical natural capital thresholds during the process of substituting natural capital for human capital (Pearce and Atkinson, 1997).

⁵ Basaigo (1995) provides an alternative classification of approaches to sustainable development. He identifies perspectives as either ecological, ethical, economic or planning, to which O'Riordan and Voisey (1998) add political science.

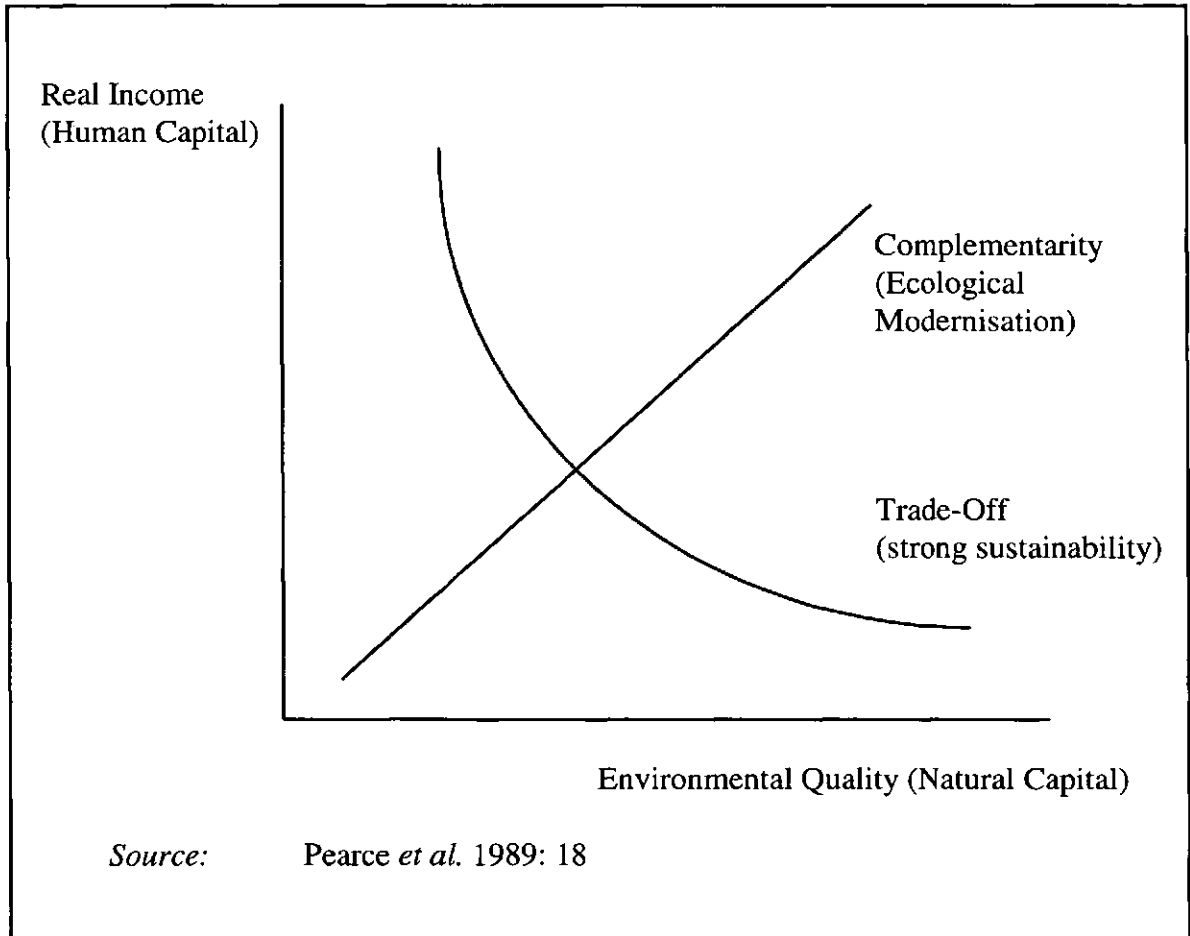
Table 2.1 Weak and Strong Sustainability

<i>Version</i>	<i>Features</i>
Very weak sustainability	Overall stock of capital assets remains stable over time, complete substitution between human and natural capital. Essential link between willingness to pay and sustainable development.
Weak sustainability	Limits set on natural capital usage. Some natural capital is critical, that is non-substitutable. Related to the precautionary principle or safe minimum standards. Trade-offs still possible.
Strong sustainability	Not all ecosystem functions and services can be adequately valued economically. Uncertainty means whatever the social benefits foregone, losses of critical natural capital are not possible.
Very strong sustainability	Steady-state economic system based on thermodynamic limits and constraints. Matter and energy throughput should be minimised.

Source: Gibbs *et al.* (1998: 1353)

By recognising the need to balance these prerogatives, weak sustainability depicts itself as a more humanitarian and realistic view of sustainable development than that held by advocates of zero growth (Auty and Brown, 1997). However, the heart of the economic standpoint is the contention that economic growth need not be sacrificed in order to achieve environmental sustainability. Instead it embraces a concept frequently termed 'ecological modernisation', the contention that entrepreneurial forces can be harnessed for environmental gain, rather than perceiving environmental protection as a brake on development or development as an inevitable source of degradation, (Nordhaus, 1992; Gouldson and Murphy, 1996). Conversely, the view of strong sustainability is that economic development and environmental protection are fundamentally irreconcilable and inevitably trade off against each other. The difference between these approaches is demonstrated in Figure 2.1.

Figure 2.1 Relationships between Environment and Development



According to the ecological modernisation critique, the central problem is that property rights over the majority of environmental resources are extremely ill defined. To illustrate, if a factory emits polluting fumes to the atmosphere (over which no defined property rights exist) there is nothing within the market system to compel the company to compensate other parties for their loss of air quality, even where tangible externalities can be readily identified (van den Bergh, 1996). Society has therefore historically under-valued and over-exploited the environment in terms of its resources and function as a pollution sink (Friedman, 1962). As industrialisation has become more widespread, this unvalued damage, or externality effect, has increased to the point where it threatens to undermine the basic viability of the economic system. The economic viewpoint contends that correcting this distortion requires the introduction of market mechanisms which re-internalise environmental externality costs until an optimal balance is achieved between human utility and natural resource conservation (Turner, 1993). Under this regime, business will be encouraged to conserve natural

resources, increase their environmental efficiency and invest in more environmentally-friendly technologies (Pearce *et al.*, 1989; Gouldson and Murphy, 1996)⁶. Furthermore, the argument also posits that ecological modernisation will stimulate new industries, reduced pollution costs and will make environment protection and economic growth complementary, the so-called 'win-win' or double dividend.

The obvious question, however, is whether there is convincing evidence to support the ecological modernisation hypothesis. As Ekins (1993) points out, the issue of sustainable development cannot be defined entirely through theoretical debate because environmental protection is ultimately an empirical question. Whilst there is evidence that reductions in resource and energy consumption have occurred in developed countries as a result of heightened environmental consciousness (Auty and Tribe, 1997), industrialisation within less developed countries (LDCs) has generally been accompanied by disproportionately high increases in pollution and waste production⁷. There is therefore considerable conjecture as to whether ecological modernisation presents a convincing approach to the environmental problem despite its acceptance by several political groupings, the EU included (Bohm, 1997).

Whilst the economic view of sustainable development is often portrayed as conceptually pragmatic but methodologically rigorous (Gibbs, 1996a; Auty and Brown, 1997), the strong sustainability vision of the ecological and equity views are seen as less precise in terms of implementing mechanisms. Instead, they stress the need for minimum ecological thresholds and a fairer economic order (Daly and Cobb, 1990; Redclift, 1992). Some proponents of very strong sustainability even reject sustainable development outright, branding it an oxymoron, a western value and a capitalist invention (see Turner, 1997), or as a fanciful notion containing moral injunctions in place of policies (see Dovers, 1997). More moderate adherents of strong sustainability have sought to question, *inter alia* whether the development or growth ethic is an appropriate response to global environmental problems, what time-scale sustainability should be planned over, or what consumption and production levels should be sustained

⁶ Environmental efficiency is defined by Pearce *et al.* (1989) as the ecological impact per unit of production or consumption. It does not, therefore, necessarily challenge the legitimacy of economic growth as a concept. Jacobs and Stott (1992) use the example of catalytic converters, which are used to make motor vehicles more environmentally benign but do not discourage their use.

⁷ Furthermore, Ekins (1993) notes that whilst energy intensity within OECD fell by 20% in the period 1973-1986, in the USA it rose again in 1987 and 1988.

(Redclift, 1987; 1992; Dahl, 1996). They contest the validity of the principle of capital substitutability by highlighting the problems associated with defining constant overall capital, a point, in fairness, acknowledged by Pearce *et. al.* (1989), especially in respect of non-renewable resources. Thus, supporters of strong sustainability reject the notion that market forces can adequately address environmental problems. Instead, they advocate an approach based on international co-operation (Redclift, 1987), the re-examination of society's ethical values (Paehlke, 1996), the democratisation or replacement of political institutions which have perpetuated the growth ethic and, more extremely, a re-examination of the resource-use policies proposed by the *Limits* model (Daly, 1992).

Many adherents to the equity vision of sustainable development view the resolution of poverty in Less-Developed Countries (LDCs) as the central policy issue (Glaeser, 1988; Redclift, 1987; Jongma, 1995). They maintain that sustainability is principally a lifestyle issue in the developed world, but that the imperatives in the LDCs are survival and the attainment of civilised levels of well-being (Castro, 1972; Redclift, 1987). As such, they are less concerned with halting economic expansion and focus more on its equitable distribution. Furthermore, they dispute whether western prescriptions of the problem and its solutions should be unquestioningly accepted (Peet and Watts, 1993; Eder, 1996).

In many ways, the equity critique expresses its disenchantment more with the political dynamics presently shaping environmental policy, which they brand as politically self-serving and 'weak', than with the underlying precept of environmental sustainability (Castro, 1972). Redclift (1987), for example, argues that sustainable development is exposed to political manipulation in the international and national arenas through policies which fail to question economic growth or which attempt to compartmentalise sustainability (Dovers, 1997). In another example, Dovers (1997) and Jewell and Steele (1996) argue that Australian and British environmental policies have sought to pursue development rather than sustainability and have attempted to assimilate it within the *status quo* rather than adopting genuine environmental ideals. Finally, Gibbs (1996) warns that (polluting) business as usual under the pretence of sustainable development may be the ultimate outcome of the weak sustainability approach. In summing up the ecocentric and equity views of strong sustainability, Redclift (1987: 36) notes:

Unless we pitch our conception of sustainable development at a level which recognises international structures, it is in danger of being yet another discarded development concept. Its polemical usefulness will have outlived its practical utility.

Though both the ecological and equity views are highly critical of the economic stance, Gibbs *et al.* (1998) suggest that the 'weak' model of sustainability may act as a Trojan horse for moves towards stronger sustainability. Conversely, they concede that strong sustainability might become drowned out within a policy-making process dominated by economic and political expediency. In response to such concerns, commentators such as O'Riordan and Voisey (1998, citing Jordan and O'Riordan, 1993) attempt to crystallise the broad policy actions required to manage the transition to strong sustainability (see Table 2.2).

To sum up, a spectrum of opinions on the nature and direction of the environmental movement and sustainable development has been expressed in the literature (O'Riordan, 1976). Whilst it is possible to argue that human society is either ecologically sustainable or not, the inevitable inclusion of social, economic and international dimensions within the debate makes agreement on either the definition of sustainable development, or the strategies required to achieve it, immensely difficult to achieve. The next section therefore explores the reasons behind the acceptance of this ambiguous concept as the prevalent political maxim for managing environmental problems and the translation of these diverse interpretations within the *realpolitik* of environmental policy.

Table 2.2 The Transition to Sustainability

	Environmental Policy	Economic Policy	Public Awareness	Public Discourse
Stage 1: Very weak sustainability	Lip service to policy integration	Minor tinkering with economic instruments on a case-by-case basis; some reinvestment of income towards the goal of sustainability	Dim awareness and little media coverage	Corporatist discussion groups and consultation exercises
Stage 2: Weak sustainability	Formal policy integration and specific targets, backed by few institutional structures	Substantial restructuring of economic incentives; large-scale reinvestment of income toward the goal of sustainability	Wider public education involving 'perforated' classroom walls	Roundtables, stakeholder group participation, and legislative surveillance
Stage 3: Strong sustainability	Binding policy integration and strong international agreements coupled to performance targets and indicators	Full valuations of the cost of living, green accounting, and creation of a 'civic income' for social use	Curriculum integration, with local educational initiatives geared to community growth	Community involvement, pairing of initiatives in the developed and developing world
Stage 4: Very strong sustainability	Strong international conventions, national duties of care, and statutory and critical support	Formal shift to sustainability accounting locally, nationally and internationally	Comprehensive cultural shift coupled with technological innovation and new community structures	Community-led initiatives become the norm

Source: O'Riordan and Voisey (1998: 16)

2.2.4 *The Political Acceptance of Sustainable Development*

Two principal reasons explain sustainable development's wide-ranging political appeal. The first is that it is a practical alternative to the zero-growth model, in that it suggests economic and ecological priorities can co-exist within a complementary framework (Gibbs, 1996a). Though the ideas underlying the Club of Rome's contentions have never been *comprehensively* disproved (or proven), implementing its recommendations would necessitate a radical and traumatic restructuring of society. Furthermore, the *Limits* prescriptions are considered by some to be ecologically Utopian and arrogant, particularly in the respect of the development needs of the LDCs (Castro, 1972; Cairncross, 1991). Sustainable development has nevertheless charted a course away from *laissez-faire* approaches to environmental management (Taylor and Buttel, 1992; Pearce *et al.*, 1993; Gonzalez, 1997). Sustainable development has also proved politically acceptable because its core themes are fluid and can be moulded to suit political exigencies (Kirkby *et al.*, 1995). Thus, sustainable development can either constitute a foundation for more reflective environmental management or, by emphasising the concept of intra-generational equity, a justification for accelerated economic expansion. This lack of clarity has led to the condemnation of sustainable development from both ends of the ecological-economic spectrum. For example, Beckerman (1995) - a staunch supporter of economic growth - regards it as a political catch-phrase and vastly over-rated, whilst Holmberg and Sandbrook (1992) - taking a more ecocentric viewpoint - fear it provides governments with a licence to pursue expansion policies. Nonetheless, Pearce *et al.* (1989) maintain that sustainable development enables the gradual and practical integration of environmental and economic priorities (also Bennett and Patel, 1993).

However, Pearce *et al.* (1990) observe that very little international action was taken between 1987 and 1990 to transform its ideas into a coherent body of policy. It was not until the United Nations Commission on Environment and Development's (UNCED) Rio Summit in 1992 that sustainable development became almost universally accepted. Turner (1997: 133) even claims:

Sustainability appears to have become the guiding principle for a global society entering the new millennium, superseding almost all others within the environment and development communities.

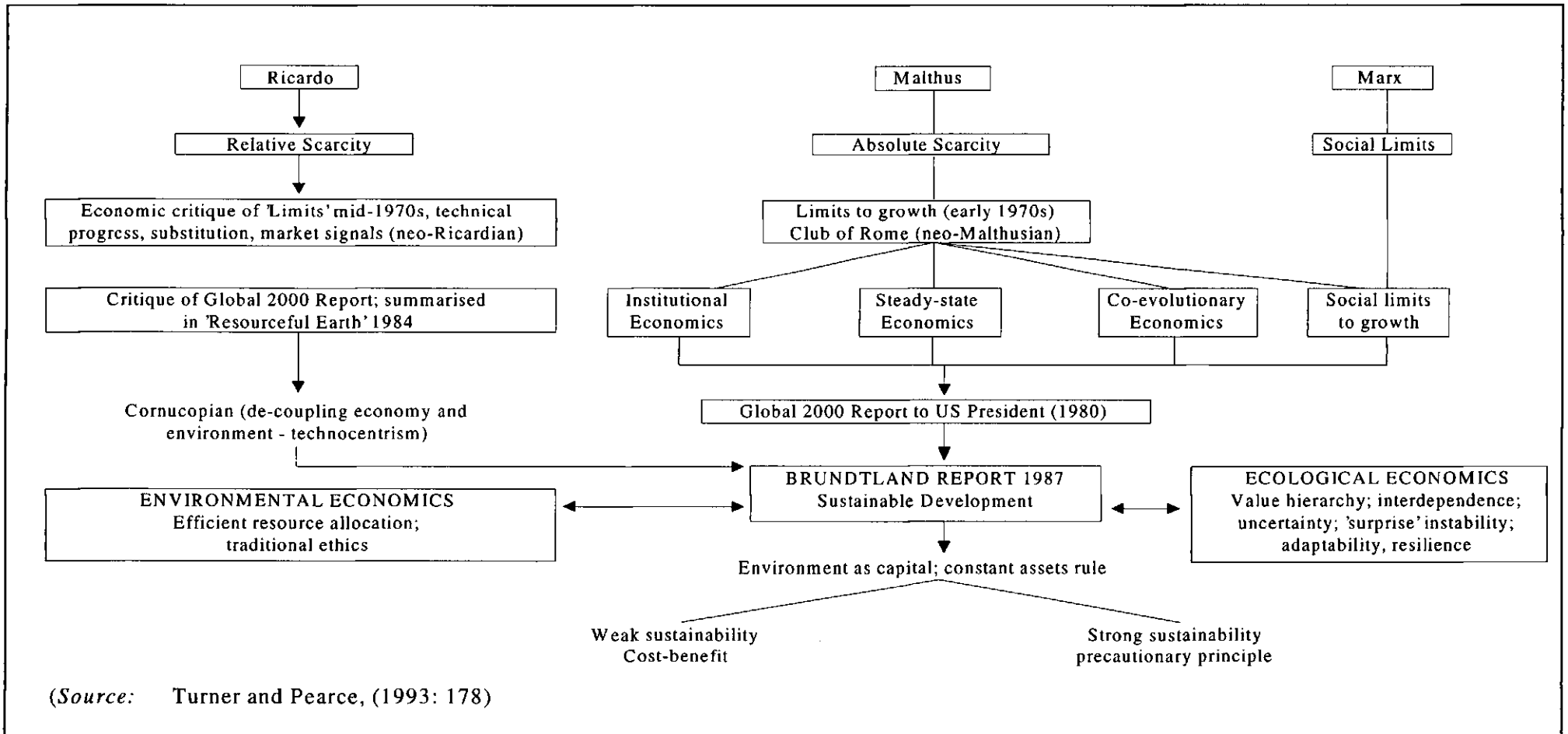
This evolution is further encapsulated in Turner and Pearce's (1993) exploration of sustainable development's influence on the academic and political agenda (Figure 2.2). However, although the preamble of the Earth Summit embraced the ecological and egalitarian aims of sustainable development, major political and ideological rifts appeared during negotiations to formulate specific policies. Some of the richer Northern nations essentially sought to preserve their economic interests by imposing stringent environmental standards globally. However, the poorer states of the Southern Hemisphere saw their development and equity as central to the debate (Middleton *et al.*, 1993; Kirkby *et al.*, 1995). Rather than accepting firm commitments to grant the South financial assistance and differentiated responsibilities - including debt relief and the licence to develop their natural resources as means of escaping poverty - some argue that the North sought to fit sustainability entirely into a context which defended their national interests (Jordan and Brown, 1997).

Furthermore, many nations have subsequently attempted to evade practical commitments in areas of high national interest, even where agreements were signed⁸. Some writers have consequently claimed that the North's chief tactic at Rio was to maintain internal sustainability by importing it from the South (Redclift, 1987; Daly and Cobb, 1990). Others contend that Rio was a severe setback for Brundtland's vision and did more to expose the problems of international co-operation within a political system based upon entrenched national interests than it did to resolve deep-seated and pressing environmental problems (Grubb *et al.*, 1993). Even prior to Rio, Johnston (1989) argued that sustainable development could never be viable on the international political agenda because it meant sacrificing national interests in the cause of global co-operation. Hurrell (1994: 16) even attacked nation states as:

Too big for the task of devising viable strategies of sustainable development which can only be developed from the bottom up ... and too small for the effective management of global problems ... which by their nature demand increasingly wide-ranging forms of international co-operation.

⁸ For example, the USA's refusal to sign the Rio Conference's Biodiversity Treaty (O'Riordan and Voisey, 1998).

Figure 2.2 Approaches to Sustainable Development



However, whilst verdicts on the success of the Earth Summit are divided, there has been almost universal disappointment at the slow pace of progress since Rio. Despite the fact that many national governments have adopted sustainable development rhetoric and local authority programmes to implement Agenda 21, the evidence suggests that far more needs to be done, both at national and local level (Gibbs *et al.*, 1998; Morphet and Hams, 1994; Agyeman *et al.*, 1996)⁹. O’Riordan and Voisey (1998) are particularly critical of the political commitment given to the environment, global partnership and welfare of the poor since the Earth Summit, as well as the reluctance of many nations to fulfil commitments they made at the Conference. Serious doubts must therefore be raised about the prospects for securing meaningful forms of sustainable development through existing international channels.

2.2.5 Summary

In summing up the sustainability debate, O’Riordan and Voisey (1998: 3) note:

Sustainability is like democracy and justice. It is a moral idea, a universally created goal to strive for, a shared basis for directing the creative and restorative energies that constitute life on Earth ... Sustainability has that ring of universal desirability about it: no one is prepared to fundamentally challenge its precepts, no matter how vague these are, simply because it is an almost holistic human wish for a viable future for this unique planet and its inhabitants. It is not surprising that sustainability, democracy and justice are seen as composite and comprehensive ideals. The grinding process of transition is of itself permanent precisely because sustainability can never be actually attained, or at least cannot be envisaged by people because of the immense and fundamental changes in our society it entails.

According to Redclift (1987: 199), however, sustainable development may be more than a mere pious hope but it is less than a rigorous methodological foundation for environmental policy. Sustainable development, he contends:

draws on two frequently opposed intellectual traditions: one concerned with the limits which nature presents to human beings, the other with the potential for human material development which is locked up in nature.

⁹ Gibbs *et al.* (1998) note that 40% of the actions agreed under the Rio conference’s Agenda 21 declaration are the responsibility of local government. Whilst this is in line with Brundtland’s vision of ‘thinking globally, acting locally,’ they stress the importance of adequate finance, authority and legislative frameworks for the development of locally-focused sustainable development programmes.

It therefore seems that the fundamental problem lies in the interpretation of sustainable development within global, national and even regional contexts rather than with the underlying concept itself. Attempts at international co-operation, arguably the most legitimate response to what are increasingly inter-related global problems, have seemingly been stifled by a political reluctance to compromise national interests. As such, the problems surrounding sustainable development span vision, commitment and methodology. Therefore, despite its acceptance as the guiding philosophy of modern environmental policy, sustainable development has yet to transform itself into a clear strategy on which to advance environmental policy into the twenty-first century.

2.3 Environmental Policy Instruments

2.3.1 Introduction

Whilst Hahn (1993) contends that by far the most common political response to environmental problems has been to ignore them, policy makers have made increasing efforts to address the complex task of identifying policy instruments that might effectively promote sustainable development. The examination of this key aspect of policy implementation begins by establishing the criteria against which policy-instrument effectiveness is gauged and by identifying and comparing the relative merits of the main policy mechanisms. Reflecting the move by many governments to incorporate price-based environmental regulation into their policy armoury, the remainder of the section examines the methodology behind price-based regulation along with its technical, political and institutional limitations.

2.3.2 Criteria for Evaluating Environmental Policy Instruments

Although much of the economics literature presents the study of environmental policy instruments as a rational science, it stands to reason that environmental economics is also influenced by the innate ambiguities of sustainable development. Helm (1998: 8) points out that 'sustainable development *could* act as a guide to policy only insofar as it can be defined, measured, and then related to actual policy decisions,' (original emphasis); similarly, policy instruments cannot be divorced from their policy context (Barbier, 1993). Segerson (1996) concludes that a distinction must therefore be made between normative and positive factors in evaluating policy instruments. The

normative approach essentially considers which instruments *should* be chosen to achieve particular objectives, whilst positive factors explain why certain approaches were *actually* adopted. This section concentrates primarily on normative factors influencing policy instrument selection but consideration is also given to positive factors in Section 2.3.6 and Chapter three.

Some of the most clearly operationalised criteria for evaluating environmental policies have been set out by European Commission (CEC, 1994) and the UK DoE (DoE, 1993). The DoE identifies seven key characteristics of an effective environmental policy instrument:

- *Environmental effectiveness* - their ability to meet policy objectives.
- *Resource costs* - the instrument that achieves the desired environmental goal at least cost to affected parties. However, the DoE recognises that the most cost-effective policy instrument does not necessarily yield the greatest environmental benefit.
- *Administrative costs* - incurred as monitoring and enforcement costs by the public sector (Heyes, 1998) and as compliance costs by the private sector.
- *Public revenues* - the effect, if any, of the policy instrument on public finances. However, the DoE acknowledges that environmental policy instruments should be designed primarily to change polluter behaviour rather than to raise revenue for the government.
- *Innovation* - the ability of the instrument to encourage innovation, lower compliance costs, or enable higher standards to be introduced in the future.
- *Competition and competitiveness* - the DoE argues that policy instruments should neither discriminate against individual companies nor produce barriers preventing new businesses from entering the market (Segerson, 1996; Bowers, 1997).
- *Fairness* - though there are inevitably winners and losers from any environmental policy, the distribution of these gains and losses must be considered as part of policy instrument selection (also Schelling, 1983).

These criteria correspond broadly to those identified by Bohm and Russell (1985), who also stress that policy instruments should remain flexible in the face of changing economic conditions. However, it is also clear that the decision-making process

becomes increasingly value-laden as more factors are considered (Hitiris, 1994). Segerson (1996) and others (Pearce *et al.*, 1989; Brisson, 1993; Rajah and Smith, 1993; Goddard, 1995) argue that environmental policy efficacy can be distilled into two basic assessment criteria, the environmental and economic effectiveness of the policy instrument. The two frameworks provided by the DoE and Segerson will be used as the main basis for assessment in this study.

2.3.3 Types of Environmental Policy Instrument

The existing literature has also identified numerous functional taxonomies of environmental policy instruments. For example, Segerson (1996) classifies mechanisms according to whether they are *ex ante* - designed to prevent environmental damage occurring - or *ex post* to correct damage already caused. Hahn (1993) differentiates between instruments which produce an explicitly defined level of environmental improvement and those that implicitly discourage pollution by imposing environmental-damage costs on polluters (see Table 2.3).

Table 2.3 Mechanisms for Environmental Control

I.	Quantity mechanisms
A.	Standards
1.	Technology-based standards
2.	Performance standards
B.	Market approaches
1.	Marketable permits
2.	Reducing market barriers
II.	Pricing Mechanisms
A.	Taxes
B.	Subsidies or subsidy elimination
C.	Marginal cost pricing in regulated industries

Source: Hahn (1993: 115)

Lévêque (1995) and Richardson (1998) make a similar distinction between instruments that produce threshold standards of improved environmental performance - broadly speaking legislative standards - and those that provide continuous incentives for improvement. Finally, Fenton and Hanley (1995) examine *where* policy instruments should be applied within the production, use and disposal cycle of products in order to achieve greatest impact. The classification used in this review divides policy mechanisms into the categories most commonly used by policy-makers; legislation, litigation, and price-based measures (DoE, 1993; Segerson, 1996), but uses the distinctions identified by other studies to help understand the functionality of each.

Legislation

The legislation approach has historically been the primary method of environmental regulation. In essence, it seeks to define clear and unambiguous environmental standards which act *ex ante* to prevent environmental damage (Segerson, 1996). Legislation can either consist of performance standards or be technology-based, for example, the specification that new motor vehicles should be designed to run on lead-free petrol (Hahn, 1993). The principal advantage of standards-based legislation is that it offers reasonable certainty as to the end result; in other words, if the only assessment criterion is the standard itself, legislation achieves a high degree of environmental efficiency¹⁰. However, its effectiveness is highly dependent on companies complying with legislation and, therefore, the legal and administrative structures which uphold the standard (DoE, 1993). Because enforcement procedures are never perfect, in practice there is invariably some 'leakage' between nominal and effective compliance (Heyes, 1998)¹¹. Lévêque (1995) identifies four variables that affect the efficacy of regulation; (i) the existence of private incentives for business to change their behaviour; (ii) the degree of informational asymmetry between regulators and regulated; (iii) the extent to which companies engage in opportunistic behaviour, such as strategic non-compliance; and (iv) the strength of government coercion. He argues that for legislation to be effective, governments must hold strong coercive powers whilst information asymmetry must be minimal in order that industry cannot conceal the extent or nature of pollution.

¹⁰ As will be argued in Chapter three, however, reliance on legislative standards (or any other policy instrument) without examining the methods by which they are achieved can lead to reductions in actual environmental standards (Turner *et al.*, 1998, Bailey, 2000).

¹¹ The UK Environment Agency estimates that the average business compliance rate with existing legislation is approximately 74% (Heyes, 1998).

Helm (1998) adds that the effectiveness of legislation is further hampered by the risk of regulatory capture in situations where information asymmetry exists. This occurs where regulators rely on industry for the data or expertise needed to define technical pollution-control standards and businesses exploit this relationship so as to influence policy in their favour (Lévêque, 1996a). This situation usually arises where regulators are significantly under-resourced in relation to their responsibilities and has frequently been observed within EU environmental policy (see Chapter three).

In terms of economic effectiveness, legislation does not impose formal charges on polluting activities, though obviously costs are incurred by industry in meeting the required standards and by public authorities in monitoring compliance (Segerson, 1996). Both parties therefore have cost-benefit judgements to make. Firms must choose whether it is cheaper to comply with or flout legislation, whilst enforcement agencies must establish an economic level of monitoring (Heyes, 1998). Although such decisions seem to compromise the notion of an environmentally-efficient policy instrument, they are inevitable in many areas of policy. Furthermore, because fines for disregarding environmental laws are usually decided by the courts rather than enforcement agencies, there is often limited scope for varying penalties to finance more rigorous monitoring regimes.

Opinion on the cost-effectiveness of legislative mechanisms is also divided. Some see legislation as unnecessarily expensive, arguing that uniform standards are insensitive to market conditions and the ability of companies to comply with legislation (Baumol and Oates, 1988; Stavins and Whitehead, 1992). Furthermore, in the absence of firm scientific evidence on the environmental degradation attributable to particular pollutants, policy-makers may impose over-burdensome standards by applying the precautionary principle (Helm, 1998). Others view it as the least cost option, however, contending that under legislative regimes businesses only pay their pollution abatement costs and not any additional incentive charge (Beder, 1996; Lévêque, 1996a). Lévêque therefore suggests that governments should focus on developing self- and joint-regulation agreements with industry in order to streamline the administration costs of environmental legislation whilst still gaining the benefits of environmentally-efficient standards (see also Whiston and Glachant, 1996).

Finally, there is substantial evidence that legislative standards have failed to reverse the general trend towards environmental degradation in America and Europe (Segerson, 1996). The argument here is that legislation can only produce threshold improvements in environmental efficiency and therefore cannot integrate environmental concerns into economic policy in the manner envisaged by Brundtland's sustainability intuition. Influenced by academic analysis highlighting the shortfalls of legislative standards, policy makers have increasingly sought to establish alternative means of expediting environmental policy, key amongst which have been the litigation and price-based approaches.

Litigation

The pursuit of companies through the courts as an *ex post* measure for neglect of their statutory environmental duties has been one of key functions of many national enforcement agencies (Segerson, 1996). However, the prosecution of illegally polluting companies by private individuals has become an increasingly prominent part of environmental policy enforcement in the USA (Heyes, 1998). Whilst no similar tradition of private litigation exists in Europe, the EU has recently signalled its intent to increase citizen access to environmental information and judicial processes with a view to encouraging more private prosecutions (CEC, 1996a).

The litigation approach has two principal merits. First, it helps to enforce existing legislation by increasing the number of unofficial pollution 'inspectors'; second, the threat of litigation costs can act as a powerful *ex ante* deterrent to would-be polluters. Against other assessment criteria, particularly those relating to the integration of environmental protection into policy and business thinking, its benefits are less clear (DoE, 1993). In addition to the fact that litigation generally only enforces standards on a 'case-by-case' basis, its environmental effectiveness and equity are heavily dependent on detecting and correctly identifying illegal polluters. Furthermore, since large companies have greater resources and expertise at their disposal than the average individual, litigation processes are heavily weighted against successful private prosecutions. Finally, legal proceedings are rarely economically efficient. Not only are the administrative costs of pursuing individual actions high, the legal system is also potentially haphazard in terms of its calculation of environmental damages where the

law is founded on precedents rather than generic principles (Segerson, 1996). For these reasons, the UK DoE has indicated that it has reservations about the widespread use of private litigation to enforce EU environmental policy (DoE, 1993; Heyes, 1998).

Price-based Mechanisms

'Price-based mechanisms' is a catch-all term encompassing a range of policy instruments which apply the Polluter Pays Principle (PPP) as a means of regulating environmental problems. Though they take various forms, each shares the common notions, first, that industrial pollution can be controlled by the re-internalisation of environmental externalities within market prices and, second, that market-based systems enable businesses to achieve cost-effective compliance with environmental standards (Baumol and Oates, 1988; Pearce *et al.*, 1989; DoE, 1993). However, because price-based measures are a form of explicit policy intervention, they do not rely on the *laissez-faire* argument that market prices will automatically ration scarce resources without the need for government intervention (Coase, 1960). Instead, price-based policies seek to harness the profit-maximisation (or loss-minimisation) self-interest of Adam Smith's market in order to achieve greater social and ecological stewardship (Schelling, 1983). Most adherents of price-based mechanisms therefore support some form of government legislation as the antecedent and underpinning of the market-based approach (Beder, 1996; Turner *et al.*, 1998)¹².

Using Hahn's (1993) classification (Table 2.3), price-based mechanisms can be divided into two categories, pricing and quantity mechanisms. The classic form of pricing mechanism is the environmental tax, where government imposes a charge on a polluting product or process designed to encourage industry to develop less damaging technologies and practices (Rajah and Smith, 1993). An alternative approach is to provide subsidies to companies that reduce their pollution, either as a simple financial 'carrot' or on the basis of deposit-refund or insurance bond systems (Hahn, 1993). Though the most common form of Pigouvian tax involves the introduction of environmental charges to achieve pre-determined legislative standards, purer forms of

¹² However, some price-based mechanisms have been successful without being underpinned by legislation. For example, prior to the forthcoming ban on leaded petrol in the EU, tax differentials between leaded and unleaded petrol encouraged most consumers and manufacturers to switch to unleaded fuel.

market-based measures introduce taxes, then leave the market to determine both the pollution level and the method by which it is achieved (Helm, 1993; 1998). Quantity mechanisms can also be used within a market-based format. The most frequent method is the tradeable permit system (Pearce *et al.*, 1995), whereby government defines desired limits for particular pollutants and either issues or sells permits to companies affected by the new standards. The theory behind this mechanism is that companies have the option to sell their permits if they can reduce their emissions. Whilst there are a number of ways tradeable permits can be organised (see Pearce and Turner, 1990; 1993; Hahn, 1989 for comprehensive accounts), the theory is that companies with lower abatement costs will sell their excess permits to companies that find pollution control cost-prohibitive. By offering the 'carrot' of a future financial reward, the tradeable permit system also encourages innovation in new technologies that can then be disseminated throughout the industry.

Because price- and market-based mechanisms operate in a variety of ways, there are numerous arguments for and against each approach. Many analysts in fact claim that a range of instruments combining legislative and other controls is essential for achieving both environmental and economic efficiency (Pearce *et al.*, 1989; Gee and von Weizsäcker, 1994). However, before the relative benefits of price-based mechanisms are reviewed, the next section reviews the theory behind the apportionment of environmental costs using price-based regulation.

2.3.4 The Economic Theory of Price-Based Policy Instruments

Environmental economists claim that the market remains the most efficient mechanism for allocating resources in the economy despite its tendency to neglect environmental externality costs. They argue, first, that incentives and decision-making are most effectively managed by individual businesses and, second, that corporate behaviour can be better controlled by changes in market prices than they can, for example, by legislation (Baumol and Oates, 1988; Hahn, 1989). If company actions can be re-directed through informed policy intervention, they maintain, the types of market failure that cause externality effects can be readily corrected. This notion was first expounded by the French economist, Pigou (1920), who researched methods of including social (including environmental) factors within the market-price system (Barbier, 1990).

The basis of much environmental economics research has therefore been the design of models seeking to establish environmentally and economically-efficient tax levels for various pollutants (see, for example, Hourcade *et al.*, 1992; Pearce *et al.*, 1989; 1993; Turner *et al.*, 1998; Xepapadeas and de Zeeuw, 1999; Gersbach and Glazer, 1999). Pearce *et al.* (1989) distil the principles underlying this technical discipline in their summary of the constituents of environmental charges (see Figure 2.3). The main principle behind all environmental valuation techniques, however, is that factors other than the simple use value of environmental resources must be included within charging mechanisms if they are to succeed in conserving natural resources for present and future generations. The literature contains a variety of methods for determining these costs, including hedonic pricing, contingent valuation and travel-cost approaches. Whilst these are not discussed in detail in this review, a footnote summary is provided and fuller accounts can be found in Pearce *et al.* (1989) and van den Bergh (1996)¹³. Pearce *et al.* (1989) recognise, however, that environmental economics experiences considerable difficulties in ascribing meaningful values to many 'softer' environmental costs, despite its methodological rigour. Costanza (1989; 1993) and More *et al.* (1996) are even more uncertain as to whether economics is able to determine the value of such ethereal factors as the existence value of environmental resources, whilst even option values for future consumption are subject to the problem of discounting¹⁴.

¹³ Pearce *et al.* (1989) review the direct, indirect, hedonic, contingent, and travel-cost approaches to environmental valuation. Direct valuation relates to the creation of a surrogate market whereby buying and selling processes attribute a value to environmental resources. Indirect valuation is based upon a 'dose-response' technique, where estimates of reduced pollution or resource consumption are calculated and adjusted to achieve an optimum balance between cost and benefit. Hedonic approaches *estimate* how much of a property differential is due to a particular environmental difference, then *infer* how much people are willing to pay for an improvement and the social value of that improvement. Contingent valuation *directly asks* people what they are prepared to pay for an environmental benefit or accept as compensation for its loss. Finally, travel-cost assesses how long and often people are prepared to pay to travel to an environmental amenity (and their duration of stay), based on an opportunity cost against the revenue gained and utility lost from going to work. Because each technique applies different valuation techniques, they are tailored towards assessments of different aspects of environmental quality.

¹⁴ The valuation of environmental resources techniques shown in the previous footnote are usually based on some reflection of human preferences. As well as considering *whether* individuals wish to use a natural resource, techniques must also consider *when* they would wish to do so. However, from the perspective of the present generation, the later a cost or benefit occurs, the less it matters. In other words, future costs or benefits are discounted. The problem for environmental economics, therefore, is to determine rates of discounting which are consistent with sustainable development's notion of leaving future generations with the same level of overall capital as the present one (Pearce *et al.*, 1989).

Figure 2.3 The Valuation of Environmental Resources

Total Economic Value = Actual Use Value + Option Value + Existence Value

where:

Option Value = Value in Use (by the individual) + Value in use by future generations + Value in use by others (vicarious value to the individual)

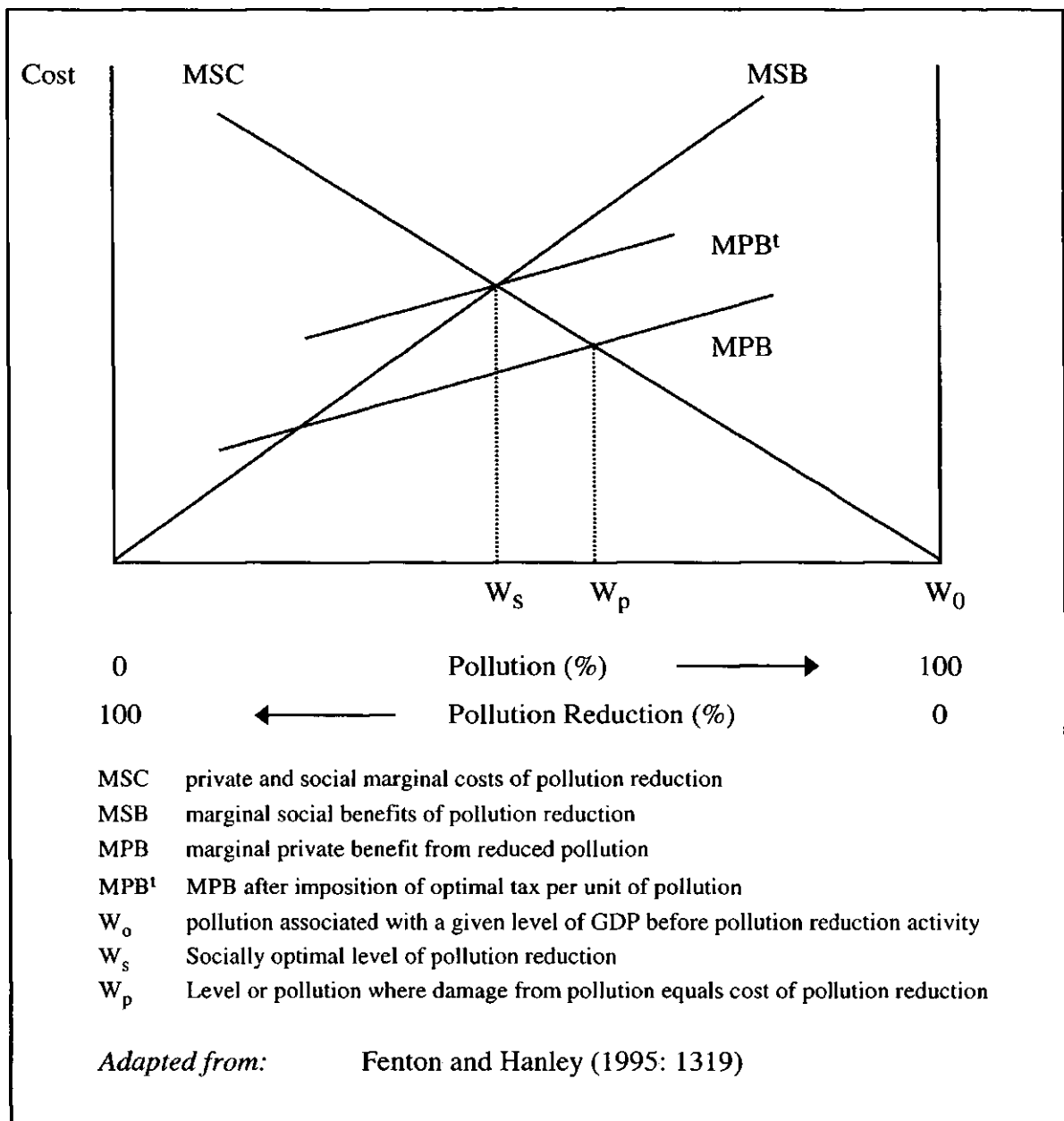
Sustainable Income is defined as Measured Income – Defensive Expenditures – Residual Pollution Value – Capital Depreciation (human and natural)

Source: Pearce *et al.* (1989: 7 and 62)

In simple terms, the application of a Pigouvian tax is designed to produce the effect exhibited in Figure 2.4. The graph shows two basic relationships; first, that as pollution reduces, the marginal benefit derived by society from the goods whose production has been forfeited (MSB) also reduces and, second, that as pollution abatement increases, so do the marginal private and social costs of abatement (MSC). Under the economic efficiency criterion, the optimal level of pollution is where any further marginal increase in pollution abatement would cost society more than it would gain (the intersect of MSC and MSB). Where a company is not exposed to any pollution costs, the market theoretically imposes no in-built restriction on the production of pollution (W_0). In practice, however, some private benefit may accrue to companies by reducing pollution to the point W_p . The aim of applying a Pigouvian tax, however, is to increase the cost of pollution from the line MPB to MPB^t , such that the company's optimal pollution level becomes W_s , the point where the marginal cost equals the marginal benefit of pollution (again, the intersect of MSC and MSB).

Three factors make any pollution abatement beyond this point uneconomic. First, it is obviously unrealistic to expect zero-pollution without returning to a pre-historic civilisation. Second, whilst many forms of pollution are socially undesirable, those addressed by price-based measures are not illegal. For unlawful pollution, environmental economists readily concede that legislative prohibition is the most appropriate course of action (Schelling, 1983).

Figure 2.4 Marginal Costs and Benefits of Pollution using Pigouvian Tax



Finally, though marginal pollution abatement costs (MSC) and benefits (MSB) are shown as straight lines in Figure 2.4, in practice, both accelerate as pollution or abatement levels increases (Beder, 1996). Therefore, based upon the criteria of maximising social welfare through natural and human capital substitution, optimal pollution, or Pareto efficiency, occurs where $MSB = MSC$. One important point must be acknowledged, however. Though some writers and policy documents refer to price-based regimes as providing *continuous* incentives for environmental improvement

(Stavins and Whitehead, 1992; CEC, 1994), the strict interpretation according to Pigouvian tax theory is that they:

effectively impose the internalisation of environmental maintenance and replacement costs [and] provide behaviourally relevant incentives to polluters and resource users in order to arrive at some allocative optimum (van den Bergh, 1996: 136-7, emphasis added).

Nonetheless, the argument that pollution taxes exert a constant pressure upon businesses to minimise their environmental cost burden cannot be entirely discounted because price-based mechanisms represent a continuous cost to industries engaged in polluting activities (Eichstädt *et al.*, 1999). Therefore, whilst the attainment of specific policy objectives will be used as the principal benchmark for evaluating the efficacy of price-based environmental policy mechanisms throughout this study, consideration will also be given to their continuous incentive effect.

2.3.5 Critique of Price-Based Policy Instruments

The principal argument in favour of price-based environmental regulation is that it is more environmentally efficient than legislative standards because polluters are made financially accountable for their actions (Brisson, 1993; CEC, 1994; Acutt and Dodgson, 1997; Turner *et al.*, 1998). As Schelling (1983: 297) puts it, 'slogans are no match for higher prices at the gas pumps.' Furthermore, because price factors rather than government legislation inform market decisions, environmental objectives can be achieved in an economically efficient manner. Whilst supporters of price-based regulation acknowledge that it does not provide a definitive answer to the thorny questions surrounding sustainable development, they maintain that environmental economics can make a substantial contribution to informing policy decisions (Bowers, 1997).

However, opponents of the environmental economics have attacked its precepts on a number of fronts. The greatest anathema to radical ecologists is the concept of environmental valuation (More *et al.*, 1996). Daly and Cobb (1990), for example, claim that the assumptions underpinning the economic valuation of environmental resources epitomise the 'misplaced concreteness' of economics. They further argue that Pigouvian taxation scenarios are constructed around a number of mythical assumptions,

the existence of perfect competition, perfect knowledge of market conditions by individual companies, and the notion that businesses and consumers respond rationally to price pressures. They assert that as these conditions rarely apply in reality - that is, positive factors distort 'rational' normative decision-making - the behavioural changes envisioned by the Pigouvian approach never materialise in practice (also Beder, 1996; Gersbach and Glazer, 1999). Finally, Daly and Cobb (1990) propose that because market systems are geared towards efficiency rather than justice (the neglect of equity) and fail to differentiate between consumers' wants and needs (the disregard of carrying capacity), they are an unsatisfactory basis for developing sustainability strategies. Costanza (1993) also notes that environmental economics can manage risks (the probability of known hazards occurring) but that it is ill-equipped to identify uncertainties (unidentified risks with unknown probabilities) within complex global ecosystems.

Other authors concede the theoretical validity of environmental valuation but highlight the practical deficiencies of the approach (Cairncross, 1991; Helm, 1998). One major concern is the difficulties surrounding the accurate monitoring of pollution and the setting of charges at levels which achieve the desired policy outcome (Defeuilley and Godard, 1997). If either of these is impracticable, then market-based mechanisms may be no more effective than government legislation (Rajah and Smith, 1993). Against this, Pearce *et al.* (1989) argue that such factors can be corrected by adopting a 'dose-response' method (also termed policy iteration), whereby environmental charges are varied until they produce their desired effect. Fenton and Hanley (1995) also point out that environmental charges have to be carefully designed to take account of the price elasticity of pollutants. Where a product is price elastic, significant reductions in demand will occur when Pigouvian taxes are levied. However, demand will be affected less for price-inelastic goods. Economic instruments may therefore not be appropriate for severe pollutants that exhibit price-inelastic behaviour if the key policy objective is to lessen their use. This means that the characteristics of each particular pollution market must be understood before price-based regulation can achieve its desired goals (Beder, 1996).

Doubts have also been expressed about the cost effectiveness of economic instruments (Rajah and Smith, 1993). Whilst both legislative and price-based regulation contain a

cost-to-industry component, those imposed by legislative regimes are comprised solely of compliance costs. By contrast, most price-based systems impose both compliance costs and an additional incentive surcharge on companies (Lévêque, 1995). Beder (1996) further claims that the apparent savings from market mechanisms in some cases merely reflect the choice by some companies to pay the charge without reducing their pollution levels. On this basis, some writers even argue that price-based mechanisms can effectively create a licence to pollute (Gibbs, 1996a; Hahn, 1993).

Finally, there is considerable debate over the distributional impact of economic instruments. If, as Friedman (1962) maintains, consumers ultimately pay all taxes in the form of higher prices, environmental charges have the potential to disproportionately affect poorer sections of the population¹⁵. Stavins and Whitehead (1992) argue this effect can be negated by ensuring environmental taxes are spent on environmental projects or to reduce taxes that are economically or socially damaging. One way this might be achieved is through reduced employment taxes (Smith, 1997)¹⁶. This is known as the concept of hypothecation or earmarking. Gee and von Weizsäcker (1994), for example, advocate an Environmental Tax Reform (ETR) package combining carbon taxes, petrol taxes, vehicle efficiency measures with levies on pesticides, nitrates and ozone depletants, to offset reductions in National Insurance and Value Added Tax, and the hypothecation of surplus revenue for property insulation, winter fuel payments and public transport provision. The practice of fiscal neutrality within environmental policy therefore seeks to address distributional issues by combining a 'double-dividend' of enhanced environmental protection and economic expansion. In other words, it attempts to operationalise the concept of ecological modernisation (Repetto *et al.*, 1992). However, Turner *et al.* (1998) note that extreme caution is required in the design and introduction of ETR if they are to avoid producing contradictory, or even damaging, incentive patterns (also Huppes *et al.*, 1992). Spackman (1997) and Mulgan (1997) further point out that, whilst the hypothecation of environmental charges might increase the transparency of public expenditure, doctrinaire hypothecation policies may distort its distribution and efficiency.

¹⁵ Pearce *et al.* (1989:7) argue, however, that businesses can only pass on a part of their pollution charge costs to customers. This issue is examined further in Chapters six and seven.

Given that sustainable development is a highly malleable concept, it is perhaps not surprising that price-based methods of environmental regulation have provoked such vigorous debate. However, whilst environmental economists have consistently championed the theoretical consistency and benefits of price-based regulation, undoubtedly the greatest difficulty is the paucity of empirical evidence to support or refute its value. This research aims to move the debate forward from its theoretical confines by providing greater understanding of the practical benefits gained from environmental taxes. However, despite the problems discussed during this review, price-based environmental regulation has begun to command greater attention from policy makers. The next section explores the reasons for this transition.

2.3.6 The Political Acceptance of Price-Based Environmental Regulation

In response to the perceived limitations of the legislative approach, US and European policy makers have increasingly begun to consider alternative methods of environmental regulation. It is therefore somewhat surprising that of the examples of price-based environmental regulation extant within the EU, few demonstrate the theoretical attributes championed by environmental economists (Huppes *et al.*, 1992; Turner *et al.*, 1998, Howe, 1996)¹⁷. The literature has devoted considerable time accounting for this apparent reticence. Evidently a number of the normative criticisms, such as the informational and price-elasticity issues raised in the previous section, have contributed towards uncertainties amongst 'vote-counting politicians' (Lélé, 1991: 613) as to whether price-based regulation represents a convincing solution to environmental problems (Helm, 1998).

For Beder (1996), however, problems of political risk dominate the discussion. After again highlighting the difficulties of translating the theoretical assertions of environmental economics into practical policies, she argues that price-based measures often fuel inflationary pressures and reinforce existing distributional inequalities. In another criticism, Richardson (1998) and Helm (1998), suggest that economic instruments do not reduce state intervention as much as might be expected, but instead

¹⁶ For example, the UK Treasury announced the introduction of the landfill tax under the slogan 'Taxing Waste not Jobs,' and announced that the revenue raised would facilitate reductions in National Insurance Contributions (Gee, 1997).

¹⁷ The most notable exceptions to this are the Swedish carbon tax and the Belgian Ecotax law (Hagengut, 1997).

actually necessitate a degree of re-regulation in order to ensure that market mechanisms operate in accordance with policy objectives. Richardson (1998) also notes that industry has proven less receptive to market-based measures than might have been expected. Whilst some industries welcome the additional flexibility they confer, many remain uneasy about the uncertainty and inflationary pressures economic instruments add to their business planning. Considering the pivotal role of industry as a formulator, facilitator and executant of environmental policy (see Chapter three), its opinions have played an important part in reinforcing political ambivalence towards economic instruments.

The adoption of price-based mechanisms has also faced a number of institutional constraints. Firstly, because environmental legislation in many countries has traditionally been underpinned by the inspection efforts of enforcement agencies, there is a strong sense that economic instruments could undermine the safeguard of active monitoring in favour of an opaque and untested system of market control (Stavins and Whitehead, 1992; Helm, 1998). Provided economic instruments deliver the required results, this should present few difficulties. If they fail, however, price-based policies and the politicians who advocate them would be extremely unpopular. Perhaps the most important institutional question, however, is at what administrative and geographical scale to apply price-based mechanisms. Goddard (1995) expresses the opinion that, since environmental issues are global symptoms of local problems, market-based instruments should be locally controlled¹⁸. However, though many European states and the USA have devolved some environmental policy-making and monitoring responsibilities, it seems unlikely they will relinquish their hold on fiscal affairs (Howe, 1996) for fear that regions may set themselves up as pollution havens in order to attract inward investment. Furthermore, because environmental problems do not respect administrative boundaries, the political accent has increasingly been on trans-boundary co-operation and the aggregation, rather than devolution, of environmental policy. Within such a climate, Skea (1995) argues that it is procedurally more simple to use legislative norms to establish national environmental policies, then experiment with economic instruments at a regional level.

¹⁸ One example of the benefits of a locally-controlled system over a national one is in the issuing of tradeable permits (DoE, 1993). If permits are issued nationally, the situation may arise where trading causes polluting activities to become regionally concentrated. Under this scenario, pollution in the worst-affected areas would become unacceptably acute even if overall national targets were achieved.

In many respects, therefore, it has proved difficult to convince political elites that the highly technical, normative and theoretical discipline of environmental economics can be translated into practical environmental strategies. Though economists have, according to Richardson (1998: 35), attempted to 'surreptitiously depoliticise environmental processes ... by focusing attention on technical questions of design and economic rationality', wider considerations, either expedient or ethical, have inevitably come to the fore. Furthermore, economic analysis has sometimes even bordered on the arrogant. For example, Goddard (1995: 194) answers concerns that consumers might not respond rationally to environmental charges, or that setting environmental charges at the 'correct' market level might not solve pollution problems, thus:

Both positions are based upon ignorance about how the market and price system work, *or can be made to work*, to allocate resource properly. Economists call this problem 'economic illiteracy' and its has bedeviled environmental policy formation for a long time. (original emphasis)

Considering that the impact of human activity on the natural environment is still poorly understood, such absolute convictions must be viewed with some scepticism. Whether others suffer from economic illiteracy or economists from environmental illiteracy is open to debate. Moreover, as Costanza (1993: 191) suggests, because 'corrosive self-interest' is at the very heart of the market system, its validity, and therefore that of economic theory, outside their specific spheres of competency cannot necessarily be assumed.

However, despite these technical, political and institutional barriers, policy makers within the EU have begun to accept that price-based regulation offers benefits which cannot be ignored (Gee, 1997). As the environmental debate has moved beyond limits to growth towards sustainable development with all its disputed meanings but undeniably wider agenda, the pressure to find policies which integrate social, economic and ecological concerns into a complementary network has intensified. Reflecting the failure of other policy instruments to achieve this integration, economic solutions are now being applied to an ever-expanding range of environmental problems. Doubts inevitably remain, however, because of the paucity of evidence of their superiority over

other policy mechanisms. However, their incorporation into the policy armoury of EU decision-makers provides an opportunity to begin addressing this deficit.

2.4 Conclusions

This chapter has reviewed two processes central to the development of environmental policy, the formulation of aims and objectives, and the design of implementing mechanisms. It has demonstrated that although sustainability issues were originally inspired by concerns about environmental degradation, ethical, political and economic issues have increasingly influenced the policy debate. Furthermore, despite attempts to develop 'objective' scientific methods for setting economic and social development within their ecological context, positive issues have again, necessarily and inevitably, emerged to shape policy decisions. A wide variety of viewpoints on the direction environmental policy should take and the methods that should be employed have been expressed in the literature. However, two key points have also emerged. First, sustainable development in a form that embraces the complementarity of economic expansion and environmental protection has established and maintained its prominence as the guiding principle for many environmental policies. Second, economic instruments are set to play an increasing role in delivering this vision because of their perceived ability to integrate environmental concerns without stifling the possibility of future economic progress.

Chapter Three

Environmental Policy-Making in the European Union

3.1 Introduction

Chapter two reviewed the concept of sustainable development and the selection of policy instruments to regulate environmental problems. However, locating any analysis entirely within the discipline of environmental economics produces only a partial understanding of the policy implementation process. This is firstly because its predictive models are based on abstracted assumptions of rational market behaviour which often translate imperfectly into the real world (see Hahn, 1989; Pearce *et al.*, 1989; 1993) and, secondly, because economics deals primarily with the effects of policies rather than their cause. As a result, this approach is 'poorly equipped to analyse the genesis of environmental regulations' (Lévêque, 1996a: 33) and the way in which political and socio-economic interests determine the objectives and shape of environmental policies (Böhmer-Christiansen, 1994). Conversely, analyses based entirely on political analysis and policy formulation (Golub, 1996; W. Wallace, 1996; Böhmer-Christiansen, 1994) do not combine their political insights with a detailed analysis of the mechanics of environmental policy implementation. Indeed, policy-makers' preoccupation with regulatory output to the neglect of implementation has been cited as a major reason for the shortfalls in EU environmental policy (Collins and Earnshaw, 1993; Demmke, 1997).

In this chapter, the key political processes affecting the implementation of EU environmental policy are discussed. The reasons for the development of the EU environmental programme are discussed, as are its legal basis, its history, and how the Union's philosophical foundations have shaped policy development. Following this, models of European integration and the key actors involved in creating and implementing environmental policy are examined along with the political pressures shaping policy development. The discussion concludes by drawing out the major political and economic issues relevant to the implementation of EU environmental policy.

The main argument proposed in this chapter is that although the authority and ambition of EU environmental policy has grown substantially in the last 25 years, it remains heavily influenced by the Union's economic, free trade and social policy agendas. As a result, progress towards sustainable development has frequently been hampered by the EU's allegiance to, amongst other things, its macro-economic objectives (Baker, 1997; Liberatore, 1997). This uncertainty, it is contended, is further fuelled by the complexity of the European polity and the fluid division of authority between the EU institutions during the formulation and implementation of environmental policy. Finally, it is argued that industrial organisations have become major players in the creation of environmental policy and that their attempts to defend vested interests sometimes undermine the ecological integrity of EU environmental initiatives. The chapter concludes that these factors have combined to produce significant dislocations between the aims and outcomes of EU environmental policy (W. Wallace, 1996).

3.2 The Development of the European Union Environmental Policy

Within the existing literature, three main reasons have been proposed to explain the emergence of the EU environmental programme; (i) the trans-national nature of environmental problems (Haigh, 1992; Kunzlik, 1994; Howe, 1996), (ii) the interdependence of economic development, ecological protection and resource management (Howe, 1996; Baker, 1997) and (iii) the maintenance of free trade within the EU Single Market (van der Stratten, 1993; Kunzlik, 1994; Moussis, 1996). Of these, the trans-national nature of environmental problems is the most ecologically located, in that it has been long recognised that pollution problems transcend national boundaries (Blacksell, 1994). Indeed, Moussis (1996) argues that the common market in pollution was established long before any international agreement on the free movement of goods. Because many environmental problems can no longer be effectively managed at national level, international co-operation and the pooling of resources have become essential components of the EU's environmental programme (Walker, 1989; Moussis, 1996; Gouldson and Murphy, 1996).

The second justification, the recognition that the EU's future economic welfare depends on its environmental stewardship, arrives at similar conclusions for different reasons. Simply put, without responsible management, economic systems run the risk of

exhausting their available supply of natural resources and pollution sinks (Jacobs, 1991). This issue is therefore at least partly one of self-preservation. Moussis (1996) develops this notion by arguing that three factors, the United Nations Conference on the Human Environment in Stockholm in 1972, the Club of Rome report (Meadows *et al.*, 1972) and student unrest in France and Germany in 1968, prompted the Member States to introduce measures addressing Europe's most pressing environmental problems. This, he suggests, was required to ward off academically-articulated demands for 'zero growth' policies within the European Economic Community (EEC). However, the EU environmental programme should not be viewed as simply a political attempt to neutralise a populist movement, but more as a realisation that economic policy could no longer neglect environmental and resource management problems (van der Straaten, 1993). If economic development is dependent on more sympathetic environmental management (Baker, 1997), then a coherent Community-wide environmental policy seemed the most logical solution (van der Straaten, 1993; Moussis, 1996). Though this justification for common European policies is primarily economic in inspiration, it concedes many of the basic precepts espoused by Hardin (1968) and the Club of Rome report (see Chapter two).

The final reason for a Community environmental policy, the protection of free trade, is more wholly allied to the EU's economic rationale and operating procedures. If individual Member States responding to domestic political pressures introduce unilateral environmental measures, any resulting differentiation in standards could potentially obstruct free trade within the Single Market (van der Straaten, 1993; Moussis, 1996). The surest means to prevent this is to harmonise legislation throughout the Member States. Though not overtly environmentalist at first glance, this approach offers several ecological benefits. First, Sbragia (1996) and Lévêque (1996a) argue that European environmental legislation is frequently driven forward by the determination of influential Member States to pursue ambitious national programmes without hampering their national industries. Whilst comparisons might be made with the 'California effect' in the USA, where America's most economically-powerful state makes the adoption of its high environmental standards a pre-condition of inter-state trade (Vogel, 1997), EU environmental policy is more accurately depicted as a struggle between this and lowest-common-denominator policy-making. This occurs where only standards acceptable to all Member States are adopted across the Union (Golub, 1996).

Notwithstanding this, the case for EU policy aggregation can also be applied to environmental protection at regional level, where sub-national administrations are more likely to succumb to local economic pressures when considering the environmental implications of new developments (Rydin, 1997). In such circumstances, policy Europeanisation helps to prevent individual regions implicitly advertising themselves as pollution 'havens'. Finally, commercial interests may also feel less threatened by intra-Community 'environmental' competition and may become more receptive to new environmental initiatives (Moussis, 1996).

Although the literature has presented both ecological and economic justifications for an EU environmental policy (O'Brien and Penna, 1997), the defence of free trade has historically played a pivotal role in policy decisions (Weale *et al.*, 1991; Bailey, 1999a). Whilst it would be misleading to suggest that the environmental programme is entirely an expedient for economic ambitions, its accent on free trade has led some theorists to accuse the EU of adopting a distorted view of sustainable development (Baker, 1997; Redclift, 1997). Chapter two argued that economic activity within sustainable societies should be organised to accommodate certain (albeit subjective) ecological limits (Pearce *et al.*, 1993; Daly and Cobb, 1990). Forrester (1999) proposes that though 'traditional' models of sustainable development recognise the interface between economic, social and ecological requirements, the environment remains largely an adjunct to the economic system (Figure 3.1). Under a systematic approach to sustainable development, however, economic and social activity must always be considered within their ecological context (Figure 3.2). Where the EU has taken the view that the Single Market takes precedence over environmental policy, it has seemingly attempted to fit ecological considerations within an economic framework. This is exemplified by the fact that, with limited exceptions such as the Danish Bottles Case (see Chapter five) and the Wallonia waste ban (Chapter eight), the vast majority of environmental legislation prohibits Member States from restricting EU free trade (Porter, 1998). Therefore, although the EU programme was a necessary response to common environmental problems, the fact that policy decisions are inevitably influenced by non-environmental priorities has meant that decision-makers are often forced to moderate straightforward environmental objectives in order to reconcile them with other policy goals.

Figure 3.1 A Traditional View of Sustainable Development Linkages

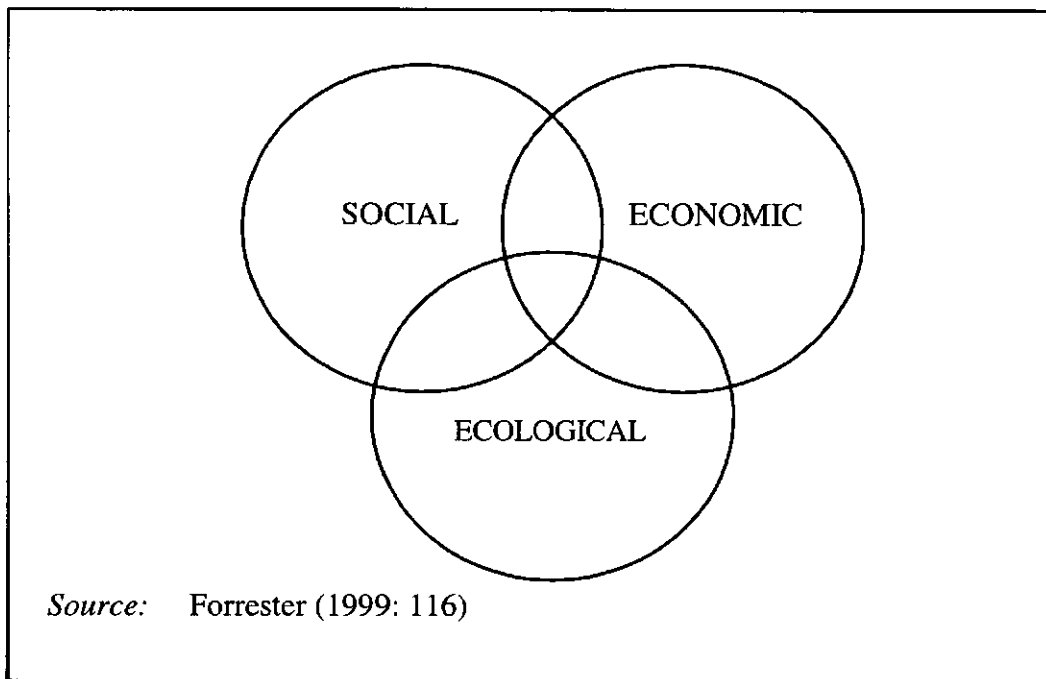
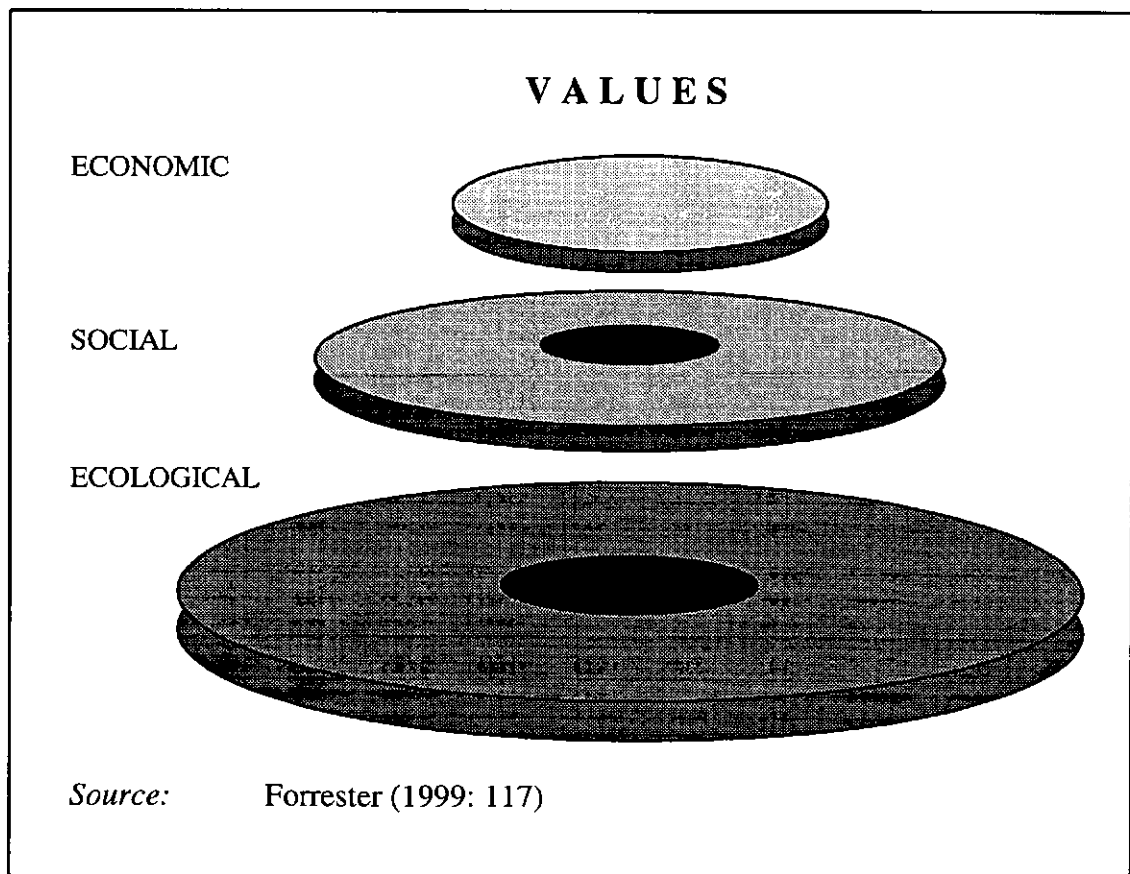


Figure 3.2 A Systematic View of Sustainable Development Linkages



3.3 The History of EU Environmental Policy

3.3.1 *Legal Basis*

Before the Single European Act (SEA) in 1986, the EEC possessed no formal jurisdiction over environmental policy. Action in this area could therefore only be justified under Article 100 of the Treaty of Rome, pertaining to the completion of the common market, or the catch-all Article 235, which permits intervention in policy areas outside the EU's official remit where this advances the overall aims of the Treaty (Baker, 1993; Vogel, 1993a; Archer and Butler, 1996). Whilst the Community's early forays into environmental policy were therefore restricted to the protection of the Common Market (Krämer, 1990), Lowe and Ward (1998a) argue that its low political profile in the 1970s enabled the EU to pursue an active agenda-setting programme. On the formal incorporation of environmental policy within European Community (EC) competencies in the SEA (W. Wallace, 1996), the principal objectives established were:

the preservation, protection and improvement of the quality of the environment, the protection of human health, and the prudent and rational utilisation of natural resources' (SEA Article 130r).

Subsequent to the SEA, therefore, new EU environmental legislation was enacted under Article 130r or, in a demonstration of the continued influence of economic and trade factors, Article 100a¹ (Kunzlik, 1994). The SEA nonetheless established environmental policy as an essential objective of the EU (Koppen, 1993), a point underlined by the prolific output in EU environmental legislation since the SEA - over 400 acts up to 1996 - (Lister, 1996) and the almost total Europeanisation of Member State environmental policies (Bennett, 1992).

The legal standing of EU environmental policy was further strengthened in the Treaty on European Union (TEU or Maastricht) in 1992, where efforts were also made to streamline environmental decision-making. Although the SEA introduced a degree of majority voting on environmental policy - prior to this, measures under Articles 100 and 235 required unanimous Council support - this was expanded under the TEU to those contained in Table 3.1.

¹ Under the Amsterdam Treaty, Article 100 became 94-5 and Articles 130s-t became 174-6. As the Packaging Directive was introduced prior to Amsterdam, the previous articles will be used in this thesis.

Table 3.1 Voting Procedures under the TEU

-
1. Qualified majority voting by Council and co-operation procedure with Parliament (under Article 130s(1)).
 2. Qualified majority voting by Council and co-decision with Parliament for internal harmonisation measures, public health and consumer protection proposals, trans-European networks, and 'general action programmes' relating to the environment (under Articles 100a, 129, 129a, 129d and 130s(3)).
 3. Unanimous voting by the Council and consultation with the Parliament (in certain cases under Article 130s(2), relating to fiscal issues, town and country planning, land use, water resource management and measures affecting Member State choice between different energy resources and the structure of its energy supply (Kunzlik, 1994: 44)).
 4. A unanimous decision by the Council under Articles 100a or 130s(2) to adopt a measure by qualified majority. This implies consultation with the Parliament. 87 votes are held within the Council, with 62 being required to support a proposal in order to achieve a qualified majority (Marks and McAdam, 1996).
-

This procedure was designed to make the adoption of environmental measures less susceptible to technical obstruction by individual Member States and, in theory, allows a more ambitious programme to be pursued than some states might otherwise sanction (Haigh, 1992; Sbragia, 1996). In order to mitigate this potential diminution of national sovereignty, the TEU also introduced the subsidiarity principle to delineate between where EU or Member-State action was most appropriate. It stipulates that the EU can only intervene in areas outside its exclusive remit where the nature and scale of required intervention means that action can be more effectively achieved at Community level. The complexities and importance of subsidiarity within environmental policy are discussed further during Section 3.4. However, the point to note at this stage is that moves towards total policy Europeanisation have not been unreservedly accepted. That subsidiarity has become a prominent part of EU environmental policy emphasises the tensions that exist between the desire for common policies and Member States' reluctance to relinquish sovereignty on matters which have a major impact on their national interests.

3.3.2 History of the EU Environmental Programme: Principles and Policies

Though the EEC possessed no legal jurisdiction over environmental policy prior to the SEA, its involvement began as early as 1972 following the United Nations Stockholm Conference. The EEC prepared the first of a series of Environmental Action Programmes (EAPs) in 1973 to establish specific policies and a more general framework in the form of guiding environmental principles (Sbragia, 1996). Those contained in the first EAP (1973-1977) included the Polluter Pays Principle (PPP), an emphasis on preventative rather than remedial action, and the inclusion of environmental considerations in all EEC decisions (Archer and Butler, 1996; Wood and Yesilada, 1996). These have been expanded over successive EAPs to include the full range of principles shown in Table 3.2. Although neither these nor the EAPs are definitive commitments to action by the Member States (Wood and Yesilada, 1996), neither are they simply pieces of rhetoric because 'they reflect certain policy priorities and in turn influence them' (Liberatore, 1997: 108). Therefore, whilst these principles fall short of a full inventory of actions necessary to achieve sustainable development, they do make a substantial contribution towards defining the characteristics of an ecologically-referenced system of environmental protection.

Table 3.2 Principles affirmed in the EU Environmental Action Programmes

-
- a. Preventing pollution at source
 - b. Incorporating environmental considerations into all planning and decision-making
 - c. Adopting the polluter-pays principle
 - d. Assessing the impact of EC policies on developing countries
 - e. Encouraging international co-operation
 - f. Promoting educational activities to increase environmental awareness
 - g. Ensuring action is taken at the most appropriate level (regional, national, EC)
 - h. Co-ordinating and harmonising the environmental programmes of individual member states
 - i. Improving the exchange of environmental information
- (Subsequent to the First Environmental Action Programme)*
- j. Ensuring that policies take a precautionary approach to environmental problems
 - k. The proximity principle (ensuring that, wherever possible, environmental damage is limited and that problems are resolved locally)
-

Adapted from: CEC (1984)

The EU's environmental principles are therefore a supplementary, if slightly subjective, tool for evaluating the contribution of environmental policies to sustainable development (Pearce *et al.*, 1993; Gibbs *et al.*, 1998). The success of environmental policies has traditionally been judged almost exclusively on the basis of whether legislative standards have been achieved. Whilst this is a quantifiable and enforceable means of evaluation, it fails to appreciate that the methods used to implement policies may have a major bearing on the overall environmental impact produced (Bailey, 2000). This could have significant implications where, for instance, Member States achieve EU legislative standards but their implementation methods actually increase the amount of environmental damage. For this reason, the EU's environmental principles will be used as part of the assessment of environmental policy success throughout this thesis².

Aside from establishing environmental principles, the First EAP focused principally on technical pollution-control standards and harmonising legislation affecting EU free trade. Many of the initiatives at this time were strongly influenced by Dutch and German legislation, as both countries had already embarked on national programmes to combat their domestic and shared pollution problems (Hildebrand, 1992). The Second EAP (1977-1981) was largely a continuation and expansion of this approach, though it did place a greater accent on international co-operation (Baker, 1997). The first material shift in emphasis came in the Third EAP (1982-1986), where the integration of environmental considerations into other policy areas became the programme's central concern (Baker, 1997). This reflected a realisation that environmental degradation could not be controlled solely through individual pollution standards since loopholes would always exist within such narrowly-focused legislation. Furthermore, it was acknowledged that EC integration had become a major influence on economic and polluting activity in Europe and, therefore, that the EU needed to recognise its responsibilities in this area (Weale and Williams, 1992).

² As a cautionary note, Lister (1996) suggests the EU environmental principles cannot easily be interpreted in case law. In particular, the preventative and precautionary principles are potentially anti-scientific, because they invite regulatory action in the absence of clear scientific evidence. However, as uncertainty and incomplete knowledge are constant themes of scientific research, it may be imprudent to base policy entirely on 'concrete' scientific knowledge (Williams and May, 1996; Costanza, 1989; Irwin, 1999).

The Fourth EAP (1987-1992) was the first since the formal incorporation of environmental policy into Community competencies and, thus, it reflected the EC's burgeoning authority in this area. Alongside the programme of specific action areas - including air, water and soil quality, waste disposal, chemical and nuclear safety and nature conservation - two more general points stand out. The first was its greater emphasis on the implementation and enforcement of EC legislation (Blacksell, 1994). Whilst the period 1989-1991 had seen an immense growth in the portfolio of EC regulation, the Commission recognised that without credible implementation and monitoring, the environmental programme would fail to achieve its key objectives. The second was the EU's acceptance of ecological modernisation, the theory that environmental protection and economic growth can be moulded into complementary objectives (CEC, 1994; Baker, 1997). In essence, this enabled the EU to contend that its programme of economic development need not be abandoned provided it was made more ecologically sensitive. This position was doubly significant in that it coincided with the 1987 Brundtland Report (WCED, 1987). Even the Maastricht Treaty used the terms *sustainable growth* and *sustainable development* almost interchangeably in an effort to make its adoption palatable to all factions within the Member States and EU institutions (Baker, 1997; Archer and Butler, 1996; Lévêque, 1996a). The EU's unwillingness to compromise its economic ambitions in order to achieve sustainable development has again led to accusations that EU environmental policy is fundamentally a weak interpretation of the concept (Turner, 1993; Gibbs *et al.*, 1998, Baker, 1997; Gibbs, 1996a).

However, it would be unfair to argue either that the Fifth EAP was not an ambitious extension of previous programmes or that the Member States were complacent in their attitude toward environmental protection. The Dobris Assessment 'The State of the Environment in the European Community' in 1992 (EC, 1992) charted a slow but relentless deterioration in Europe's environmental quality. Accepting the challenges laid down by the Dobris Assessment, the Fifth EAP acknowledged that existing policies had failed to deal adequately with environmental problems caused by EU integration (CEC, 1992a). It went on to state that:

the achievement of the programme and its objective of sustainable development constitutes one of the major political and economic challenges for the Community [and]...constitutes a major turning point (CEC, 1992a: 145).

The Fifth EAP therefore sought to develop a more holistic strategy for EU environmental policy based on the concept of sustainable development. It charted six areas of environmental degradation where urgent action was required; the management of natural resources, integrated pollution control and waste management, reduced consumption of non-renewable energy, improved mobility management, environmental quality in urban areas and the improvement of public health and safety. Additionally, five sectors of the economy were targeted as the focus of attention, industry, energy, transport, agriculture and tourism (Blacksell, 1994).

The Fifth EAP also recognised the need to incorporate a broader range of policy instruments into the EU armoury (CEC, 1992a). Previously the EU and its Member States had regulated environmental problems almost entirely through legislative standards. The Fifth EAP responded to the perceived failure of this approach (as expressed in the Dobris Assessment) by proposing that economic instruments and voluntary agreements with industry should play an enhanced role in Member-State strategies (see Chapter 2) (CEC, 1996b; 1997a; 1997b). However, whilst the EU's support for a more dynamic and, arguably, less autocratic approach to environmental policy was undoubtedly a serious attempt to realise Brundtland's vision of sustainable development, the Fifth EAP continued to adhere to the notion that free trade and economic expansion were compatible with, or even essential to, sustainable development (Lélé, 1991). Furthermore, the EU's adherence to ecological modernisation should not be underestimated in terms of its policy implications.

In addition, though the Fourth and Fifth EAPs both recognised the importance of policy implementation, the general consensus is that a fundamental gap still exists between the aims of the environmental programme and the practical results achieved (Collins and Earnshaw, 1993; Demmke, 1997; CEC, 1998a). The EU has therefore introduced a series of initiatives to facilitate the implementation process. Two of the most important have been the 'LIFE'³ project and the European Environment Agency (EEA). LIFE is a financial instrument designed to assist the development and implementation of environmental policy through the funding of research and practical projects (Sharp, 1998). Schemes funded under LIFE have particularly focused on nature conservation

³ L'Instrument Financier pour l'Environnement, established under Council Regulation 1973/92.

under the Habitats and Species Directive⁴ but have been less extensively employed to deal with waste management, as numerous national schemes already exist⁵. The EEA was formally established in 1990 to succeed the Commission's first environmental information programme, CORINE⁶. It differs markedly from national environment agencies in that its function is 'largely informational rather than regulatory or implementational' (Lister, 1996: 15). However, its purpose also includes the promotion of effective environmental protection through the provision of a common information system for harmonising EU standards (Wynne and Waterton, 1998). Although the Commission has considered extending the EEA's powers to those of an international enforcement agency, this has been opposed by some Member States as an unacceptable intrusion into national affairs (Clinton-Davis, 1992; Macrory, 1992). Despite the Commission's long-standing concerns over the 'implementation deficit' of environmental policy, both implementation and monitoring remain predominantly controlled by the Member States. Whilst this helps to bolster the credibility of local enforcement systems, it has also produced a sometimes disjointed process. Furthermore, because the Commission has little administrative machinery or access rights to verify Member-State implementation reports, the existing system has proven difficult to police effectively at EU level (Demmke, 1997).

The Fifth EAP also established three *ad hoc* dialogue groups to advise the Commission and help reduce the implementation deficit; the General Consultative Forum, the Network for the Implementation and Enforcement of Community Law (IMPEL), and the Environmental Policy Review Group (Kunzlik, 1994; Moussis, 1996). These groups, whose compositions and remits are shown in Table 3.3, function to assist structured information exchange, orderly interest representation and the fostering of common approaches to environmental policy implementation. Despite these measures, the implementation of environmental policies remains largely outside the EU's remit, whilst the problems of effective enforcement are, if anything, becoming more acute (CEC, 1998a). This issue is discussed further in Section 3.4.3.

⁴ Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora, 92/43/EEC.

⁵ For example, the ENTRUST scheme used to divert funds from the UK's landfill tax towards research on ameliorating the environmental impact of landfilling.

⁶ Council Regulation (EEC) 1210/90. CORINE (Co-ordination of Information on the Environment), which was managed within the Environment Directorate between 1985 and 1991.

Table 3.3 Dialogue Groups provided by the Fifth Environmental Programme

<i>Group</i>	<i>Composition</i>	<i>Function</i>
The General Consultative Forum	Business, organised labour, professional groups, NGOs, local and regional government, the Commission	To represent interest groups in a structured manner in the discussion of new policy initiatives and their implementation
The Network for the Implementation and Enforcement of Community Law	National and Commission officials involved in the implementation of EU environmental measures	Exchange of information and the development of common approaches to practical problems concerning the implementation of EU environmental measures
The Environmental Policy Review Group	Senior officials from the Member States and the Commission	To develop mutual understanding and exchange of views on environmental policy and measures

Adapted from: Kunzlik (1994: 82)

In summary, the EU environmental programme has been transformed in the past 25 years from a narrow body of pollution-control legislation into a core EU policy (Lowe and Ward, 1998a; Maloney and Richardson, 1994). It has made increasingly clear commitments to integrated environmental management and sustainable development (Rydin, 1997) and has established judicious principles upon which to found its policy involvement. However, two less encouraging themes have also become apparent. First, the EU has made only limited concessions to its commitment to economic expansion and free trade and, therefore, matching the rhetoric of environmental integration with practical success has been extremely difficult (Howe, 1996; Gibbs, 1996b). Second, poor implementation of environmental policies has been a constant feature of the EAPs. Collins and Earnshaw (1993) and Demmke (1997) note that even well-designed policies will fail to control environmental degradation if credible implementation and enforcement procedures do not exist. Notwithstanding the conflicts between economic and environmental priorities within the EU system, the efficacy of the environmental programme is dependent on the existence of effective policy-making and implementation structures. The following section considers the key forces and actors shaping these processes.

3.4 The EU Environmental Policy Process: Integration, Actors and Procedures

Since the early days of the EEC there has been intense academic and political interest in the nature of European integration (see, for example, Archer and Butler, 1996; Balassa, 1961; Bulmer, 1983; Moravcsik, 1991; Schmitter, 1996; Slater, 1982). Though this debate covers all EU activities and has been transformed by the successive treaties, Weale (1996) argues that EU integration has a particular relevance for environmental policy. This is principally because environmental problems are inherently international in character and therefore warrant supranational responses (the first justification for EU involvement in environmental policy). Environmental policy therefore amplifies key issues concerning international co-operation and the development of common policies within the EU (Howe, 1996; Weale, 1996). This discussion of the policy-making process begins with a brief review of the main models of European integration and the key players involved in the environmental policy process. It then explores three themes critical to EU environmental policy, the demarcation of responsibilities between the EU and the Member States under the subsidiarity principle, the distinctive character of EU policy-making, and issues concerning the formulation, transposition and enforcement of EU legislation.

3.4.1 Models of European Integration

Whilst federalism and nationalism constitute the polar opposites of inter-state relations, few would argue that either has entirely dominated EU politics (Wise and Gibb, 1993). However, because federalist and nationalist tendencies regularly emerge within EU policy-making, it is important to appreciate how they are manifested within the EU polity. Federalism in its purest form favours the creation of a Community founded on a strong constitutional and institutional framework, wherein formal and mutually-agreed divisions of jurisdiction exist between central and regional government (Wise and Gibb, 1993; McDonald, 1999). Whilst there are obviously many forms this can take, in most cases, federalism implies the existence of a central executive body which holds legislative authority over its regional constituents in agreed policy areas. Nationalism may also take many forms but, generally speaking, it either totally opposes integration or insists that the nation state must be the primary focus of all government activity. Under this form of governance all international relations are managed on an inter-governmental basis.

Three principal models of integration have been used to explain the interplay between nationalism and federalism in the EU. The first is confederalism, a tempered form of federalism that advocates a more gradual and partial power-sharing process. Under confederal systems, the 'central' governing body is rarely an entirely separate entity but, rather, is comprised of representatives from constituent regional executives convening to make decisions on specific, mutually-agreed policy areas (Wise and Gibb, 1993). However, the confederal model is substantively more than a series of *ad hoc* international agreements, since each member of the confederation is bound by formal treaties which commit them to common aims and policy-making within specified areas. From this, it follows that EU law holds a higher position than national law within areas covered by the treaties and is directly applicable in the Member States (Archer and Butler, 1996). The operation of this hierarchy in the EU is explored further in Section 3.4.3.

Functionalism proposes a modified version of the confederal model based on the development of common policies in specific areas where mutual advantage can be gained from co-operation. Functional alliances form where national policy-makers agree that certain policy areas can be managed more effectively on a collective basis, either because of economies of scale or, in the case of the environment, in response to trans-national problems (Long and Ashworth, 1999). Under functionalist arrangements, the tendency is for state governments to retain responsibility for nationally-sensitive policy areas, such as security, defence and foreign policy, but to seek greater integration in less contentious areas, notably trade (Balassa, 1961; McDonald, 1999). As with confederalism, functionalist models depend on the voluntary but formal concession of selected powers to a 'central' decision-making process (Haas, 1964). Archer and Butler (1996) argue that a variant of this, neo-functionalism, best characterised European affairs from the 1950s to the mid-1970s. The neo-functionalists contended that as functional areas became officially integrated, a spill-over process would occur where political and economic elites transferred their key loyalties, expectations and goals from the national to the EU arena (McDonald, 1999). Whilst this occurred to a degree and may even be accelerating with the advent of the Single Currency, there is little evidence that a consistent switch away from state-centred bargaining has occurred.

Intergovernmentalism proposes a contrasting vision of European integration, which claims that EU decision-making is typified by voluntary co-operative agreements between independent states. Supporters of this approach argue that EU integration should only be extended beyond inter-governmental contact where the creation of supranational bodies creates significant additional benefits for all participants (McDonald, 1999). Policy negotiations under inter-governmentalism are therefore typified by the defence of national interests and the form of lowest-common-denominator bargaining found in many international treaties. Despite the increased integration engendered by the SEA, TEU and the Amsterdam Treaty, Slater (1982) argues that inter-governmentalism has always formed a significant component of EU political relations. Moravcsik (1991: 216) further remarks that:

From its inception, the EC has been based on interstate bargainings between its leading Member States ... each government views the EC through the lens of its own policy preferences; EC politics is the continuation of domestic policies by other means.

Against this, Zito (1999) argues that certain characteristics of EU behaviour, such as the ability of majority voting to move the EU beyond lowest-common-denominator decision modes, can encourage a more entrepreneurial style of policy advocacy amongst some EU actors. Probably no single existing model can fully describe the complex and evolving politics of the EU. Instead features of each ephemerally characterise EU relations then are superseded as political circumstances, issues and personnel change (Wise and Gibb, 1993). As de Tocqueville (cited in Höreth, 1999: 249), notes:

A new form of government has been found which is neither precisely national nor federal ... and the new word to express this new thing does not yet exist.

What is evident, however, is that tensions between national interests and collective action have a major bearing on the development and implementation of EU environmental policies. Whilst institutional, ideational and interest-led pressures are almost inevitable within any political grouping, environmental policy more than most demands unified action (Weale, 1996). Thus, the decision-making behaviour of the EU Member States has a crucial bearing on environmental policy outcome and, consequently, progress towards sustainable development. The question must therefore be whether the EU's complex and multi-layered administrative machinery possesses

sufficient cohesion that it can develop environmental policies consistent with the aims of sustainable development. In order to assess this issue, it is useful to examine the roles of key actors in the policy process.

3.4.2 The Key Actors

The Commission

The first role of the Commission, as guardian of the EU Treaties, is to propose legislation promoting the Union's aims of economic and political integration. These proposals are based on the Commission's interpretation of the Treaties and specific EU action programmes (Wise and Gibb, 1993). It is also largely responsible for monitoring the transposition and compliance with EU law by the Member States (Cowgill, 1992). However, its remit excludes any formal executive role, either in the acceptance of legislation - this remains the domain of the Council of Ministers and, more recently, the European Parliament (EP) - or the practical implementation of EU policies (H. Wallace, 1996a; 1996b; Lévêque, 1996a). The Commission instead functions principally to initiate policy and, in conjunction with the European Court of Justice, to defend EU law (Howe, 1996; Wendon, 1998). Golub (1996) argues, however, that the Commission's right and duty to introduce proposals which advance EU integration enables it to exert considerable influence over policy agendas in the Member States.

The Commission is divided into Directorates-General covering specific areas of policy (Sbragia, 1996). Each Directorate-General is then divided into a number of policy domains; the Environment Directorate includes units covering, for example, Integration Policy and Environmental Instruments, Environment Quality and Natural Resources, and Industry and the Environment. Since the Commission's responsibilities are divided between many specialist departments, proposals emanating from one unit unavoidably impact on the work of other sections and directorates. For example, environmental legislation routinely affects industry and therefore has implications for the Internal Market and Competition Directorates (Collins and Earnshaw, 1993). The problem is therefore to resolve not only incompatibilities between new legislation and the EU Treaties, but also conflicts between different divisions of the Commission. Whilst inter-departmental tensions pervade all forms of government, a distinctive, piecemeal and exceptionally legalistic style of policy-making has emerged in the EU because

legislation must be co-ordinated within the Commission even before it is debated by the EP and the Council of Ministers (H. Wallace, 1996a; Metcalfe, 1992). This, critics argue, has a particularly strong impact on EU environmental policy since the tendency is to produce legislation which is legally consistent but fails to translate the EU's expansive visions of sustainable development into effective programmes of action (Collins and Earnshaw, 1993; Demmke, 1997).

Although the Commission plays a prominent role in formulating legislation affecting the citizens of Europe, its staff are appointed by Member-State governments rather than being directly elected representatives. A number of authors have argued that this constitutes a major democratic deficit, since major policy responsibilities have been entrusted in the Commission without it being subject to commensurate public accountability (Collins and Earnshaw, 1993; Wood and Yesilada, 1996; van der Straaten, 1993). Under EU procedures, democratic sanction can only be imposed on the Commission by the European Parliament (see section on the EP) or by the refusal of Member States to support the re-nomination of individual commissioners. Whilst this arrangement is designed to maintain the independence of the Commission and the balance of power between institutions, some commentators have proposed that it encourages the Commission to be aloof and out of touch with public opinion (H. Wallace, 1996a; W. Wallace, 1996). Even disregarding these accusations, the Commission's 'top-down' style of policy-making is seemingly at odds with the locally-focused politics advocated by the Brundtland Commission as the foundation for sustainable development. Nonetheless, by virtue of its place as a main initiator of EU legislation, the Commission remains a key institution, both in terms of European integration and improved environmental protection.

*The Council of Ministers*⁷

The Council of Ministers is the main executive body of the EU's ordinary agenda. It exists essentially to set and prioritise policy issues and to decide upon Commission proposals (Cowgill, 1992; Barnes and Barnes, 1999)⁸. The Council is even less of a single entity than the Commission but is instead 'a revolving group consisting of the relevant ministers from each of the Member States who meet periodically to decide upon Commission proposals which fall within their jurisdiction' (Lister, 1996: 15). Therefore, though the politically non-partisan but pro-integration Commission is responsible for proposing measures to implement the EU agenda, the combined national governments hold the definitive reins on power (H. Wallace, 1996a)⁹. Golub (1996) clarifies this distinction by demarcating between the extent to which each institution holds *influence* or *power* over EU decision-making. Although the Commission has considerable influence over the policy agenda because of its right to propose legislation, this cannot be equated to the exercise of power since this is retained by the Member States through the Council of Ministers. However, because the agenda is defined at least partially extraneously from the Member States, this prevents the EU agenda being entirely sequestered by national interests (also H. Wallace, 1996a).

Although this separation of duties is necessary to avoid the over-concentration of authority in one EU institution, it inevitably creates a degree of tension between the Commission and the Council. Whilst the Commission has the duty to pursue EU integration, more disparate views on specific policies and the general direction of integration are inevitably articulated within the Council (Bulmer, 1983; Slater, 1982; Pfander, 1996). This friction between policy 'proposers' and 'deciders' is particularly pronounced in environmental policy because the Commission and the EP have traditionally been more sympathetic to the environmental lobby than the Council (Sbragia, 1996; Lévêque, 1996b). It is also important to recognise that strategic

⁷ Reference is made to 'the Council' to describe the Council of Ministers (renamed the Council of the European Union in the Amsterdam Treaty). The term Council of Ministers will be used in the thesis, as this was its official title during the negotiation of the Packaging Waste Directive. This definition of the Council should not be confused with meetings of EU heads of state acting as the European Council.

⁸ However, it should be noted that the authority of the European Council supersedes that of the Council of Ministers in matters of overall EU strategy (the extra-ordinary agenda), (Wood and Yesilada, 1996). In fact, because both institutions are composed of national government representatives, major conflicts between the two groups are rarely made public.

⁹ Though note the new powers held by the European Parliament under the co-decision procedure (see Section 3.4.2.3).

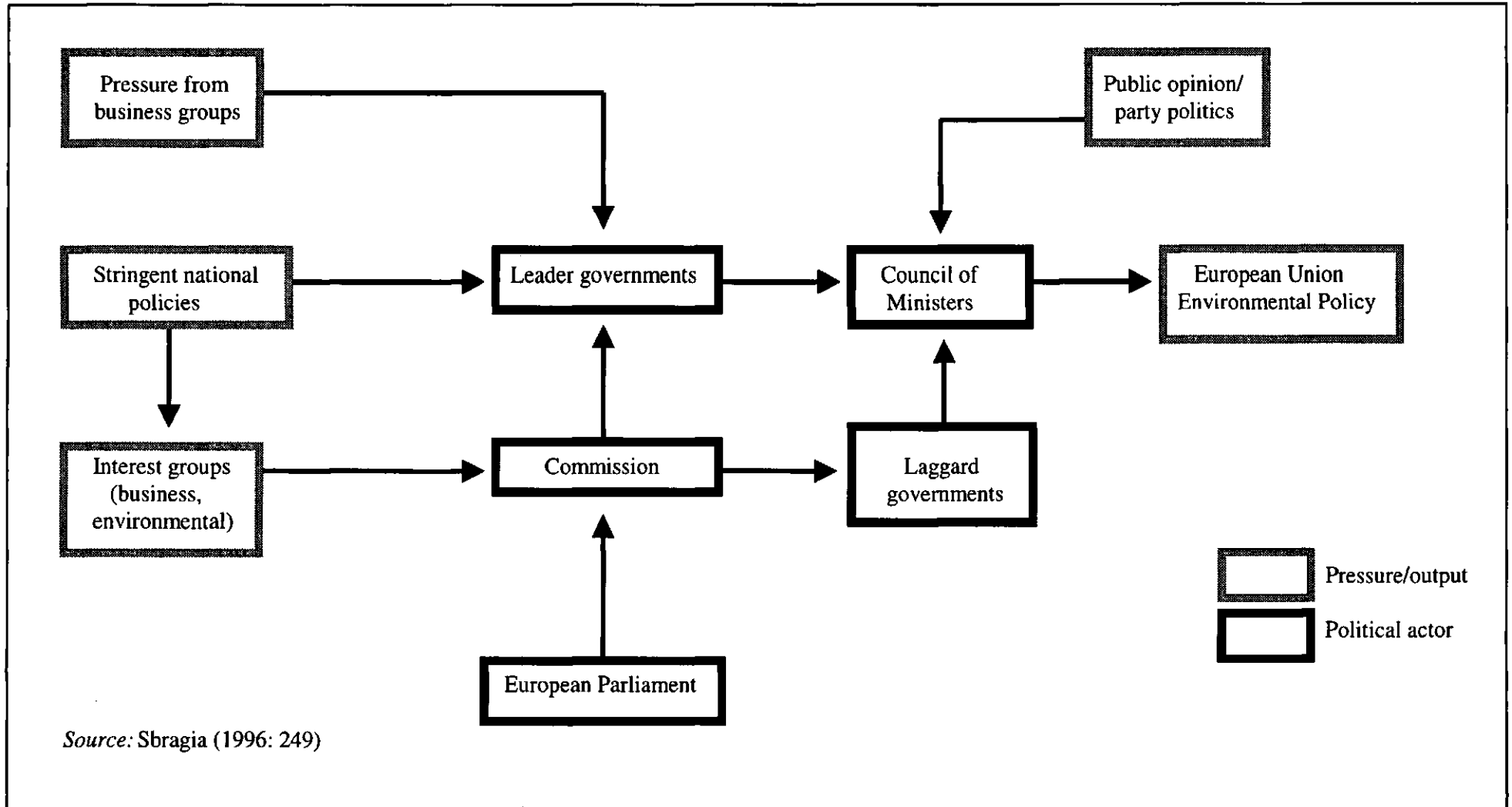
negotiating is commonplace in the Council, in that Member States often lend support to proposals they do not fully support in exchange for concessions in areas where they have particular national interests (Lévêque, 1996b). A key example of such conflicts relates to policies on air pollution, where the UK is an avid proponent of air quality standards on the grounds they are more economic to monitor, whilst Germany supports emissions control standards, since it considers them to be the surest means of protecting German forests from airborne pollution (Hajer, 1995). In another example, Germany has consistently pushed for the use of 'Best Available Technique' (BAT) approaches to pollution control, whereas Britain supports the more pragmatic BATNEEC (Best Available Technique *Not Entailing Excessive Cost*) approach (Skea and Smith, 1998). Prior to the SEA, when the majority of environmental policy decisions required unanimous Council approval, individual states were able to disable decision-making expediently by exercising their veto power¹⁰. Though qualified majority-voting procedures partly circumvent such obstructionist tactics, Collins and Earnshaw (1993: 225) note:

Despite member states' articulation in Council of deeply entrenched preferences based on national circumstances and practices, negotiation in Council remains best characterised as a search for consensus...This search for unanimity...increases the possibility that EC environmental legislation will be vague, ambiguous and sometimes superficial.

The expression of national interests and the Council's desire to maintain a unified front on key issues have been constant features of EU policy. Whether entrepreneurial or 'lowest-common-denominator' policies are adopted depends on who holds the majority at the Council veto point (Golub, 1996). Sbragia (1996) therefore argues that EU environmental policy is epitomised by a 'push-pull' dynamic between environmental 'leader' and 'laggard' states (see Figure 3.3). She contends that the advancement of EU environmental policies often relies on the introduction of national legislation by leader states such as Germany, the Netherlands and the Nordic countries.

¹⁰ Though Sbragia (1996) also notes that environmental ministers enjoy higher prestige within the Council than they do domestically and therefore can gain greater credence for their positions by demonstrating unanimous support for substantive policy advances.

Figure 3.3 The Push-Pull Dynamic of EU Environmental Policy-Making



This is not always because of environmental zeal, however, as having one's national administrative procedures adopted across the EU also reduces the disruption and expense of complying with EU legislation (Barnes and Barnes, 1999). When faced with a policy 'push', the EU must either challenge national legislation, incorporate it within EU law, or allow a potential market distortion to occur. Whether or not the standard is adopted depends on the leader state's ability to defend its legislation against the counter-arguments of environmental 'laggard' states. Sbragia argues that the cycle then begins again, gradually strengthening the influence of EU environmental policy. However, though this process increases environmental standards incrementally, the dynamic is often a cumbersome and piece-meal way of conducting environmental policy. Weale (1996) also notes that the push-pull system is less oriented towards a problem-based approach to sustainable development than it is to managing the democratic intricacies of the EU's complex political make-up.

The European Parliament

Two principal official powers are conferred on the EP under the Maastricht Treaty; the ability to propose and veto amendments to EU acts (Articles 189(b) and 189(c)) and the right, under Article 144, to dismiss the Commission by two-thirds majority for failure to fulfil its statutory roles (Cowgill, 1992). However, although it is the only EU institution directly elected by its citizens, Parliament's influence has historically been quite marginal. Weale (1999: 45) notes that:

Regarded from the point of view of parliamentary systems in Europe, the powers of the European Parliament appear few. It is not the formal source of legislation. It does not appoint or overthrow governments. Its party alignments are not well established. It is less attractive than national parliaments to those for whom politics is a career rather than a form of early retirement. It does not have the last say on legislative matters. In short, it still has to make the transition fully from a consultative body to a legislative body holding the executive to account.

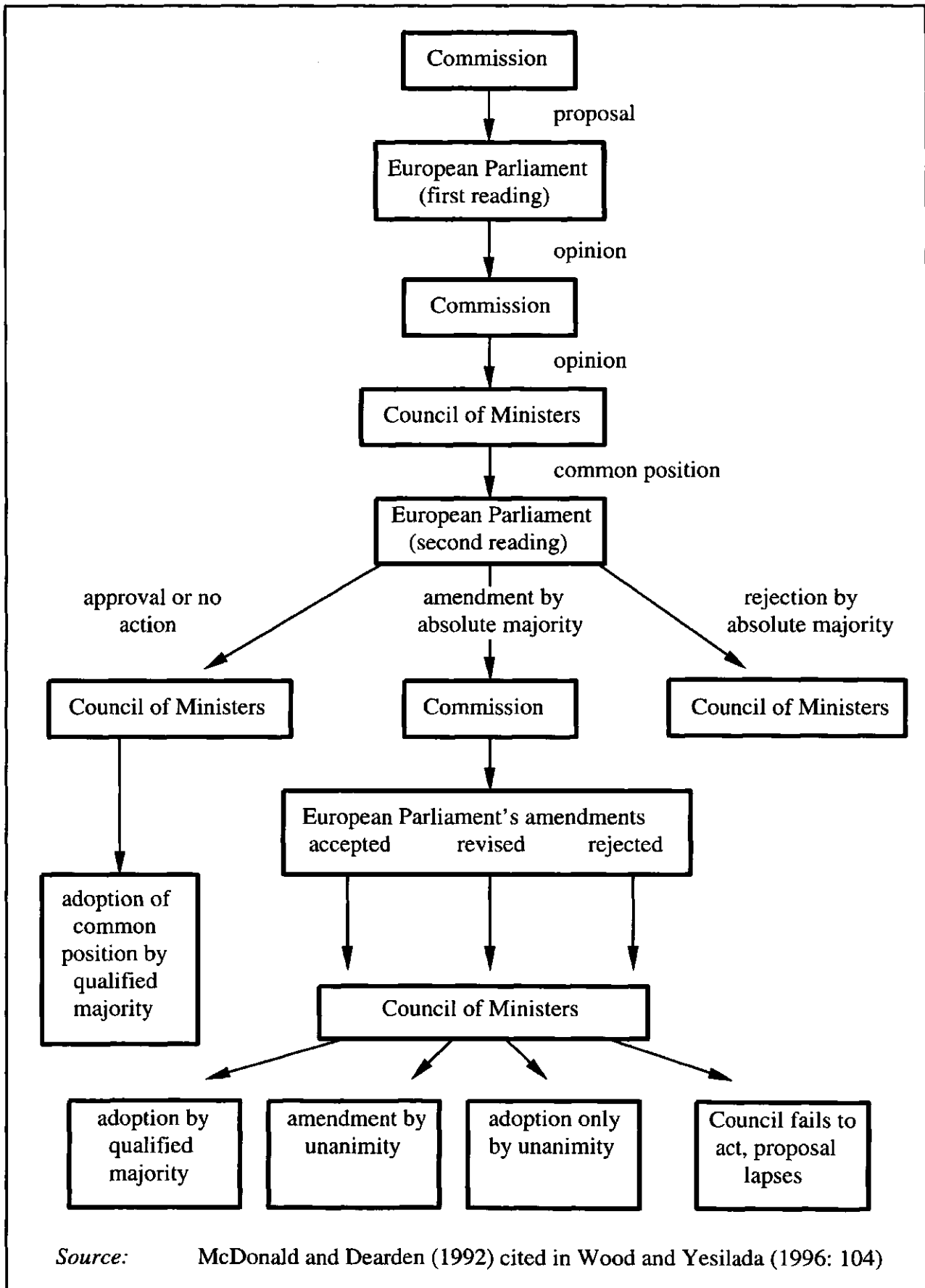
Indeed, its role was almost entirely consultative prior to the SEA and TEU. Under the consultation procedure, EC legislation was given a single reading in Parliament for the proposal of amendments but neither the Council nor Commission were obliged to accept Parliamentary suggestions. An EP amendment accepted by the Commission could be passed by qualified majority in Council, but those rejected required unanimous support in the Council in order to be included in legislation. The Parliament retained

the right under this procedure to issue an official opinion on the final legislation (Wood and Yesilada, 1996: 103). This situation further fuelled the accusation that the EU was fundamentally a democratically-deficient body (van der Straaten, 1993; Wood and Yesilada, 1996).

The co-operation and co-decision procedures have reinforced the EP's meaningful involvement in policy decisions (see Figures 3.4 and 3.5, also Wood and Yesilada, 1996 for a comprehensive account of these processes). Under the co-operation procedure, legislation rejected by the Parliament on a second reading can only become law if the Council unanimously over-rides Parliament's veto. In the co-decision procedure, a conciliation committee is formed if the Council and Parliament fail to agree a proposal. As either party may reject the proposed solution, Wood and Yesilada (1996) argue this effectively makes Parliament a co-equal legislative body in policy areas falling within this procedure.

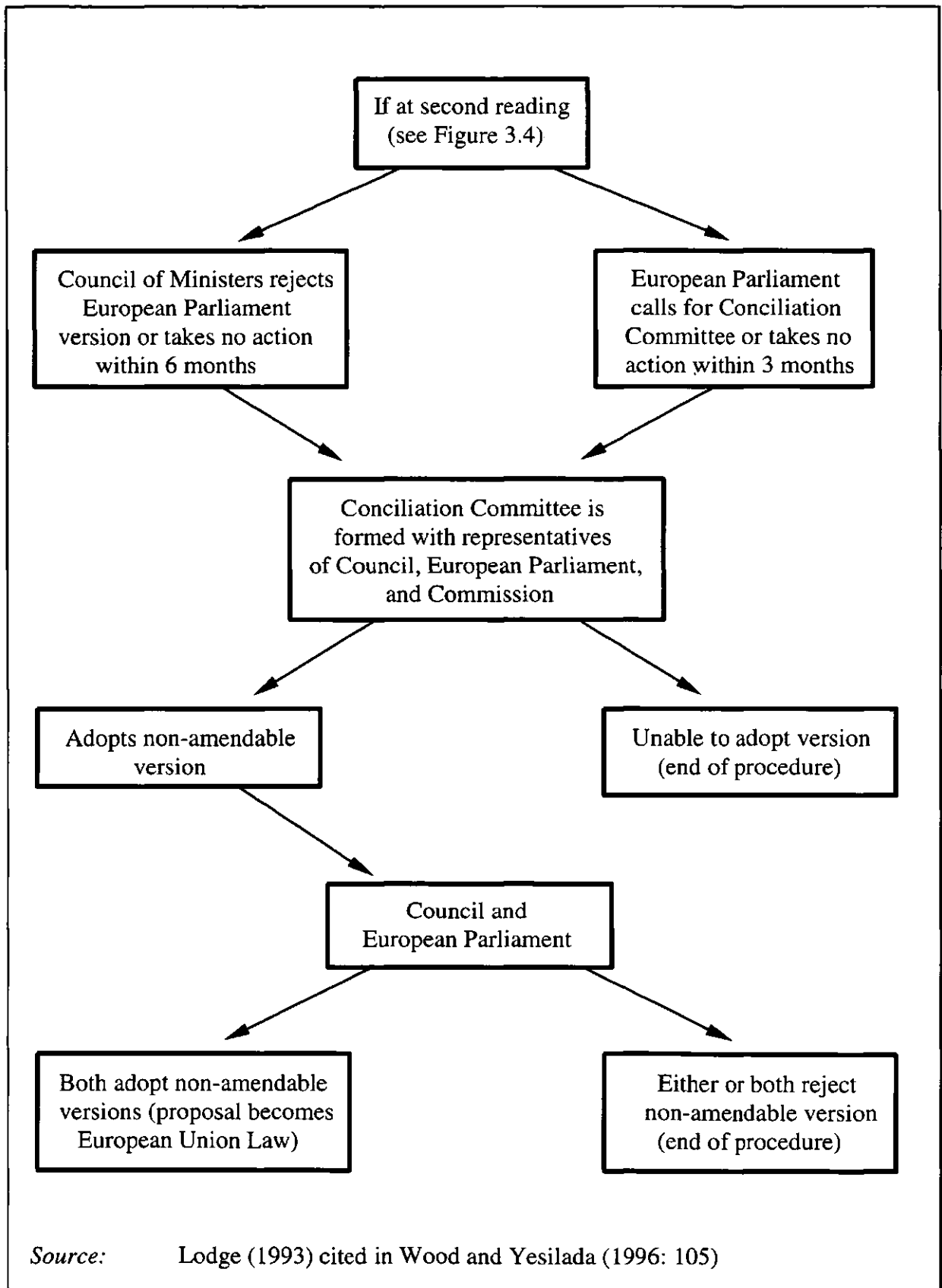
The EP's decision-making powers have been further strengthened by Article 251 of the Amsterdam Treaty, which expands the range of policies falling within the co-decision procedure (see Chapter nine). Some commentators nonetheless maintain that policy decisions are predominantly taken at some distance from direct democratic scrutiny (Wood and Yesilada, 1996; Tsoukalis, 1997). Sbragia (1996) notes, however, that the Parliament has assumed a particularly active role in environmental policy, partly because of its strong 'Green' contingent and partly since legislation enacted under Article 100a now automatically triggers the co-operation procedure. This forces the Council to accept EP amendments if it does not wish to see environmental initiatives fail entirely (Weale, 1999). The new procedures have therefore created an avenue whereby the European Parliament can extend its influence on environmental policy decisions beyond those customary for a national parliament (Lévêque, 1996a; Weale, 1999). Moreover, they have increased the number of institutions and interests with significant influence over the policy-making process (Zito, 2000). However, it will probably be some time before the full impact of the Amsterdam Treaty on EU decision-making procedures is fully clear.

Figure 3.4 The Co-operation Procedure



Source: McDonald and Dearden (1992) cited in Wood and Yesilada (1996: 104)

Figure 3.5 The Co-decision Procedure



Non-Government Actors

While the EU institutions and Member States comprise the formal actors in environmental policy-making, the literature also recognises the influence of non-government interest representation groups (Lévêque, 1996a). Many environmental and other agencies operate under this banner (see Marks and McAdam (1996) for a detailed discussion); however, in line with the focus of the research, this section concentrates on industry's role within policy formulation and implementation. A number of authors have documented the rise of corporate lobbying at the EU (Salisbury, 1984; McLaughlin *et al.*, 1993; McLaughlin and Greenwood, 1995; McAleavey and Mitchell, 1994; Mazey and Richardson, 1993; Schmitter and Streeck 1991a, 1991b; Lévêque, 1996a) and logically equated this with the process of policy Europeanisation (Marks and McAdam, 1996). Tsoukalis (1997) even suggests that economic and capital globalisation have intensified industry's influence over policy formulation as environmental self-determination has shifted away from the nation state, (McGrew, 1993; Bailey, 1999b).

Corporate ventures into the EU arena generally take two forms; businesses can either oppose legislation that threatens their profitability or strategically support measures which offer potential competitive benefits (O'Brien and Penna, 1997). As such, both the *absolute* gains and losses of environmental legislation and their *relative* distribution between competing companies are important determinants of corporate response (Lévêque, 1996a). This does not mean that businesses have the capability to block environmental legislation entirely - this would require appreciable Member-State or EU institutional support - in practice, they are more likely to adopt strategies which minimise their absolute losses and maximise relative gains (van der Straaten, 1993).

Lévêque (1996a) argues that, in pursuit of competitive gains, sequences of engagement frequently emerge in the corporate lobbying process. Those companies threatened with the greatest absolute losses from an environmental initiative, or with the most sophisticated monitoring and campaigning networks (generally speaking, large and multi-national companies), make the first attempt to influence policy-makers. This alerts other major companies, which realise they might become absolute losers should the first entrants secure competitive advantage. Thus, successive waves of corporate lobbying are created. By contrast, the views of the Small-Medium Enterprise (SME)

sector rarely gain a full hearing because of their lack of effective lobbying networks. Lévêque's typology is framed around a divisive competitive ethic but Jacquemin and Wright (1994) observe that industries faced with common commercial threats (as often occurs with environmental legislation) tend to form issue or sector coalitions. Corporate lobbying is therefore essentially a pragmatic and issue-based process, rather than one framed around permanent actors fulfilling rigidly defined roles. Woolcock *et al.* (1991) and Egan (1997) also point out that national and cultural differences further complicate this mosaic. For instance, German industry has often demonstrated a willingness to accept new social responsibilities provided they are introduced in a manner which does not disrupt competition (a relationship with government sometimes referred to as *Ordnungspolitik*), whereas British business tends to maintain a short-termist and financially-oriented view (Egan, 1997).

Finally, the intensity of industry lobbying is also affected by the 'price elasticity' of products or services under policy scrutiny (Lévêque, 1996a) (see Chapter two). When faced with the prospect of shouldering additional environmental costs for a price-elastic product, businesses will campaign intensively against new regulation. Conversely, industry may be more receptive to legislation where environmental costs can be readily recouped. In most cases, therefore, industries will adopt the least *overall* cost response to environmental regulation. Determining the trade-offs is therefore often a complex procedure. Companies must assess, first, whether it will be more expensive to absorb the costs of new regulation or mount an obstructive campaign and, second, the publicity benefits accruing from co-operation with policy-makers (Smith, 1993; Welford and Prescott, 1994). However, industry's response to environmental regulation remains essentially interest-led and utilitarian in character.

Considering the impression given thus far, that industry habitually obstructs EU environmental policy, it is worth noting the reasons why policy-makers have elicited industry participation to the extent they have. The most obvious explanation comes from the Fifth EAP's recognition that legalistic, top-down policies failed to arrest environmental decline in Europe and therefore that more inclusive approaches were needed (CEC, 1992a; 1996c). In the globalising economy, business is both part of the environmental problem and an important element of its resolution (Hawken, 1993). A number of writers (McLaughlin *et al.*, 1993; McLaughlin and Greenwood, 1995; Haas,

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1999; Mazey and Richardson, 1992) have added that industry has been co-opted into policy decisions for less high-minded reasons. They highlight that many EU institutions possess insufficient expertise to define technically-complex pollution legislation (van der Straaten, 1993) and are therefore reliant on industry for information and advice. This is particularly the case with the Environment Directorate, which is widely renowned for being under-resourced and low in expertise in relation to the magnitude of its tasks (Aguilar Fernández, 1994; Baker, 1997; Redmond, 1996). There is the danger, however, that within such an informationally-asymmetric relationship the Commission may be susceptible to regulatory capture (Kunzlik, 1994; Aguilar Fernández, 1994). This can occur in two forms; companies can either use scientific results to add authority to their viewpoints and impress politicians with apparently 'hard' facts (Funcowitz and Raveltz, 1990), or they can exploit scientific uncertainties to dispute the environmental risks associated with certain industrial processes (Underdahl, 1990). Under these conditions it is conceivable that large businesses will succeed in influencing policy-decisions significantly in their favour.

Lévêque (1996a) also identifies that industry, with the assent of the Commission, has become increasingly instrumental in determining the methods by which environmental policies are implemented. Here he identifies that the 'top-down' approach of early EAPs is being gradually augmented by (i) self-regulation, whereby industries voluntarily agree to control certain practices in order to stave off restrictive legislation and (ii) co-regulation, wherein a broad regulatory framework is established but industries retain considerable flexibility in defining how environmental targets are met. The most prominent examples of voluntary regulation in EU environmental policy are the Eco-labelling¹¹ and Eco-Management and Audit Schemes (EMAS)¹² (Welford, 1995; Johnson and Corcelle, 1995; DoE, 1995). By contrast, the co-regulation option best describes the Commission's commitment to price- and market-based environmental policies in the Fifth EAP (see Chapter five). Lévêque (1995) argues, however, that self-regulation might theoretically help in devolving environmental stewardship duties to industry but, in practice, it tends to be ineffective. This, he

¹¹ Council Regulation (EEC) 880/92, relating to the Life Cycle Assessment (LCA) and labelling of product groups. The first product group to go under the EU's LCA hammer was washing machines.

¹² Council Regulation (EEC) 1836/93, which defines a framework and requirements for business-led environmental management systems. Companies meeting these obtain accreditation to the EU EMAS standard, which can then be used to market the environmental credentials of the business.

contends, is because self-regulation is dependent on government's ability to regulate industrial activities through the threat of future legislation and the existence of positive market gains to industry from self-regulation (see Chapter two). If companies either 'free-ride' voluntary regulation or pursue competitive gains in a manner that undermines the regulation, self-regulatory agreements will fail to achieve their policy goals (Bailey, 1999b; 2000; Whiston and Glachant, 1996; Segerson and Micelli, 1998). Lévêque (1996b) therefore supports co-regulation based upon clear objectives and operating rules as a more practicable way of nurturing effective industry initiatives. While schemes such as Eco-labelling and EMAS continue to be supported by the Commission, price-based methods of co-regulation are finding greater favour with EU policy-makers (CEC, 1998a).

Whilst it is apparent that industry's involvement in environmental regulation has become more intense with policy Europeanisation, its engagement has never been straightforward. This is understandable, as there are no simple formulae for reconciling industry's commercial interests with the more eclectic aims of sustainable development. Industry has often been a dissenting voice against environmental policies it views as impractical or economically-damaging but it has simultaneously forged partnerships with policy-makers to help craft innovative methods of policy implementation. Aside from its functional alliances to defend vested interests, the tactical and partisan behaviour of industry defies simple and neat classification. What is clear, however, is that EU policy-makers have needed to scrutinise industry's engagement in the policy process in order to maintain an equitable balance between economic and environmental objectives (Beder, 1996; Bailey, 2000). Nonetheless, it is inconceivable that industry should, or even could, be excluded from participating in the formulation of environmental policy if the programme is to achieve credible environmental integration.

3.4.3 Decision-Making in the EU: Issues and Processes

Justifying EU Intervention: The Subsidiarity Principle

As the debate on models of European integration has shown, decisions on whether action should be taken at European-Union or Member-State level are an important element of the complex confederalist-intergovernmental persona of the EU. Whilst some Member States might welcome greater co-operation and federalisation, the near

rejection of the Maastricht Treaty in the Danish referendum emphasised the extent of political and public uncertainty over how far EU integration should be permitted to extend (Scott *et al.*, 1994). Furthermore, the political debate in the UK, particularly within the Conservative party, suggests that influential political factions fear that creeping federalism is already occurring in the EU (Höffe, 1996).

Although the balance between federalism and inter-governmentalism is partly maintained by the segregation of duties between EU institutions, general guidelines are still required on the sanctioning of EU intervention. The main framework for this is the subsidiarity principle, which Jacques Delors' Commission championed as the basis for all EU policies during the ratification of the Maastricht Treaty (Barnes and Barnes, 1999). It states that:

In areas which do not fall within its exclusive competence, the Community shall take action ... only if and in so far as the objectives of the proposed action cannot be sufficiently achieved by the Member States and can therefore, by reason of scale or effects ... be better achieved by the Community (quoted in Toth, 1994: 268).

In part, subsidiarity was designed to allay fears of a federal EU hegemony by placing the burden of proof on the Commission to demonstrate the need for EU action (Barnes and Barnes, 1999). Kunzlik (1994) agrees that subsidiarity requires the Commission to work within its powers and proportionately according to need. Against this, it is couched in such imprecise language that it simultaneously provides criteria by which pro-integration states might argue for increased EU intervention. Its clearest strength is therefore that its general framework allows decisions to be made on a flexible case-by-case basis. Van Kersbergen and Verbeeck (1994: 220) nonetheless make the caustic observation that:

The adoption of subsidiarity was cheered by both defendants of more authority at the Community level, like France and Germany, and opponents of such a development, as, for instance, the United Kingdom. Not surprisingly, subsidiarity rapidly became 'the Euroconcept all can admire by giving it the meaning they want.'

Even the most cursory scrutiny reveals the principle's extreme vagueness. Green (1994), for instance, argues that there are few objective means of deciding whether a state can 'sufficiently' resolve a problem, whether solutions can be 'better' achieved by

the EU, or what scale of Community intervention is most appropriate in any particular case. Toth (1994) further argues that subsidiarity is virtually unjusticiable by the ECJ without it becoming embroiled in political rather than judicial decisions - a view shared by the House of Lords Select Committee on the European Communities (House of Lords, 1996).

It has nonetheless been suggested that subsidiarity might be used to manage the allocation of environmental responsibilities between national and regional government (Scott *et al.*, 1994). By doing this, subsidiarity could help reduce the democratic deficit in the EU (Stoker, 1991; Conzelmann, 1995) and, specifically within environmental policy, it could enable locally-led solutions to be implemented within framework EU policies. This, it is suggested, provides an appropriate interpretation of the 'think globally, act locally' strategy envisaged by the Rio Conference's Local Agenda 21 programme (Gibbs *et al.*, 1996; Local Government Management Board (LGMB), 1994). However, a number of authors are sceptical as to the extent to which subsidiarity will filter down to sub-national government (Scott *et al.*, 1994; Green, 1994). It is therefore more accurate to view it as a general guide to policy decisions than as an exact route-map for regulating European integration (Höffe, 1994).

Subsidiarity has consequently become as much a symbol of the problems associated with governance in the EU as it has the solution to the allocation of policy responsibilities. For some commentators, subsidiarity, or a mechanism serving similar functions, is an essential pre-requisite of good governance in the EU (Blackhurst, 1994). Others maintain that it has merely smoothed over ideological differences between Member States 'by being so vague and insubstantial as to allow all parties to believe that it is furthering their cause, while in reality furthering none' (Green, 1994: 298). Moreover, Demmke (1997: 65) claims that subsidiarity might theoretically encourage flexible environmental governance but it does little to resolve material policy problems:

A far greater service would certainly be rendered to the cause of environmental protection if instead of indulging in ideological disputes about interpretation of the subsidiarity principle the public debate concentrated much more on the serious causes of the shortfalls in implementation and enforcement and discussed the necessary reform of the environmental authorities and of environmental legislation.

Demmke therefore highlights the point that whilst mechanisms akin to subsidiarity are necessary in most federal constitutions, they are principally devices for managing trans-national politics. Subsidiarity is neither a complete solution to confederalist-intergovernmental tensions nor a recipe for promoting sustainable development in the EU. According to Barnes and Barnes (1999), greater clarity in its application and its linkages with the principles of sustainable development are the highest priorities for the future.

Agreeing Environmental Legislation

Whilst interpreting the subsidiarity principle is a contentious issue in its own right, tensions between the EU's environmental 'leader' and 'laggard' states truly come to the fore in the formulation of environmental legislation. During these negotiations, the standards adopted as EU law depend principally on, first, the Commission's interpretation of the Treaties, second, the degree of in-built excess in Commission proposals in the knowledge they will be negotiated down (Golub, 1996) and, third, the balance of power within the Council on any particular issue (H. Wallace, 1996a). Weale (1996) describes the resulting policy-making structure as a system of *concurrent majorities*. He rejects either the notion that there is a dominant coalition of Member States which consistently imposes its will on the minority or the idea of a random 'merry-go-round' of individual countries grabbing the environmental policy agenda. Instead he suggests that veto or obstructive power is sufficiently well distributed between EU policy-making bodies that agreement amongst a wide range of actors is required before policies can be adopted. Since the issues and interests change with each environmental initiative, this precludes either dominant majorities or random opportunistic policy-making. With the advent of the co-decision procedure and the expansion of majority voting, Weale's typology may now serve as a more complete descriptor of EU decision-making than Collins and Earnshaw's (1993) notion of consensual bargaining in the Council¹³. Instead, legislation must be made acceptable to a sufficiently large majority of policy actors whose positions are informed, at least in part, by industry and environmentalist interest groups. Inevitably this leads to a process

¹³ The Europa website notes, however, that only 14% of Council decisions are typically made by qualified majority (<http://ue.eu.int/en/Info/index.htm>).

founded on negotiation and compromise, the tendency of which is to make final legislation 'the aggregated and transformed standards of their original champions modified under the need to secure political accommodation from powerful veto players' (Weale, 1996: 607). The fundamental problem with this decision-making structure, according to Weale, is that it encourages a bargaining mentality within policy negotiations rather than a focus on 'objective' problem solving. There is therefore an implicit conflict between the first justification for EU environmental policy, the need for international co-operation to combat trans-boundary degradation, and the *praxis* of decision-making within the confederalist-intergovernmental framework.

Even though environmental initiatives are not always weakened by the need to obtain concurrent majorities, the existing literature suggests this is more often than not the case (Golub, 1996; Goodman, 1996; Bailey, 1999a)¹⁴. The articulation of these dynamics in the Packaging Directive is explored further in Chapter five. However, the question is whether this interest-led process of negotiation and compromise - which is prominent within but not unique to either environmental policy or the EU - is capable of promoting sustainable development effectively (Weale, 1996). Weale, amongst others (Demmke, 1994; 1997; Chayes and Chayes, 1993; Krämer, 1996), suggests that it is not and cites the 'implementation deficit' within the Member States as evidence of the problems stemming from this dynamic. That said, it is easier to criticise the system's obvious failings than it is to propose a system of policy formulation and implementation which can promote sustainable development whilst simultaneously preserving democracy and national sovereignty within the EU.

3.4.4 Policy-making and Enforcement Procedures in the EU

Before proposals for environmental legislation can become EU law, they must pass through a series of consultation stages. Panels of experts within the Environment Directorate first discuss each proposal in order to assess its practicability. Following this, the proposal is forwarded to the college of Commissioners in order that its compatibility with the work of other Directorates can be assessed (Krämer, 1990). This process of co-ordinating the work of the Commission can take several months or even

¹⁴ Weale (1996) cites the Integrated Pollution Prevention and Control Directive (96/61/EC) as an example of EU policy upgrading national legislation. The UK attempted to have its national legislation adopted as EU law but the final directive became far broader in scope than the UK originally intended.

years (Demmke, 1997), suggesting that Weale's concept of concurrent majorities also extends to intra-institutional negotiations. Once outstanding issues have been resolved, the proposal is formally adopted by the Commission and sent to the Council of Ministers, their respective civil services, the EP and the Economic and Social Committee¹⁵. Objections or amendments from the Member States are then fed back to the Committee of Permanent Representatives (COREPER)¹⁶, whose role includes assisting negotiations on the finer points of each proposal. At this stage, the co-operation and co-decision procedures (Figures 3.4 and 3.5) may be triggered to resolve disputes between the Council of Ministers and the EP (Barnes and Barnes, 1999). Should these stages be successfully completed, the proposal is accepted by the Council and passed to a series of 'comitology' committees which handle the technicalities of reconciling EU with national law (Demmke, 1997)¹⁷. If the requisite majority is impossible to reach, the proposal is either returned to the Commission for revision or abandoned (Westlake, 1995; Wood and Yesilada, 1996).

EU legislation can be enacted in many forms but the two principal types of policy are regulations or directives. Regulations are, in effect, direct transpositions of EU law and are immediately applicable within the Member States without national legislative action (Lister, 1996). Directives, on the other hand, are binding upon the Member States in terms of the obligation to act and the standards to be achieved but not the legislative format or implementation methods employed (Pfander, 1996; Krämer, 1991). Therefore, whilst regulations might be seen as vehicles of a federalist policy style, directives are more consistent with the confederal approach¹⁸. To date, the majority of environmental legislation has been enacted in the form of directives. Three reasons can be proposed to explain this. First, directives are preferred particularly by Member States that are reluctant to sanction the transfer of legislative activity to the EU. Second, because directives permit greater implementation flexibility, they are more

¹⁵ The European Economic and Social Committee (ESC) is an advisory body to the EU decision-making institutions. It is comprised of representatives from employers, trade unions, small business, farmers' associations and the professions and functions to add greater democracy and consensus to the decision-making process.

¹⁶ Each Member State has a permanent representation to the EU in Brussels, directed by an ambassador, called a permanent representative. The tasks of COREPER Committees include the preparation of Council discussions and texts for legislative adoption.

¹⁷ There are currently 31 comitology committees working in the field of environmental policy, of which 21 are specifically concerned with legislative affairs.

¹⁸ This distinction is not universal, however. Howe (1996) argues that even the use of directives is considered too centralised and prescriptive in the USA, despite its more federalised system.

sensitive to local political, planning and economic conditions (Collins and Earnshaw, 1993). This is relevant to all states but is particularly pertinent for those with federal political systems, such as Germany and Belgium, where many environmental policy functions are carried out by regional government (Demmke, 1997). Finally, most directives are designed in a manner which permits Member States to introduce higher standards than those contained in EU legislation, provided their measures neither impede free trade nor prevent other states from complying with EU law. This enables environmental 'leader' states to respond to domestic pressures whilst also, importantly, maintaining momentum behind the push-pull dynamic (Krämer, 1991). However, because directives entitle Member States to employ their preferred methods of implementation so long as legal minima are met, they are undeniably more complicated to monitor against Single Market requirements. Furthermore, if as was suggested previously, implementation methodology is a major determinant of the overall impact of environmental legislation, it is uncertain whether the flexibility inherent in directives promotes either uniform or sustainable environmental standards throughout the EU (Bailey, 1999a).

Following final acceptance of an environmental directive, Member States are required to transpose EU legislation and notify the Commission of their compliance measures. There then follows a transition period before full implementation is required¹⁹. Should a Member State fail to transpose or properly implement a directive, first responsibility for enforcing EU law falls upon the Commission (Article 169) (Kunzlik, 1994). Initially this takes the form of bilateral exchanges with the Member State in order to resolve outstanding problems without recourse to formal proceedings. Should this fail, the Commission informs the state in a '169-letter' that it believes a failure to fulfil Treaty obligations has taken place. The letter also specifies a time period within which the state's observations are required (Collins and Earnshaw, 1993). Though relatively few proceedings progress beyond this point, the Commission may issue a 'reasoned opinion' if it is not satisfied with the state's reply. This customarily sets out the reasons why the state's justifications are not accepted and a timeframe for compliance. Where a Member State persists with a transgression, proceedings may then be initiated with the

¹⁹ This period can be extremely protracted. For example, the Packaging Directive was adopted in 1994 but full implementation is not required until 2001. These delays exist ostensibly to allow Member States time to agree legislation and implementation plans but, as a corollary, creates a lengthy delay between the initiation of EU legislation and its standards being enforced.

ECJ, whose role is to act as final arbiters of the dispute (Cowgill, 1992). Whilst receiving a 169 letter is normally enough to shame a Member State into action (Chayes and Chayes, 1993), Collins and Earnshaw (1993) and Demmke (1997) both argue that the number of infringements of EU environmental policy is increasing and that the enforcement procedure is too time-consuming to deal with them effectively. Ken Collins, former Chairman of the European Parliament Committee on the Environment, reported to the UK House of Lords as far back as 1992 that: 'We have now reached the stage where if we do not tackle implementation and enforcement properly, there seems very little point in producing new environmental law' (House of Lords, 1992: para. 67). More recently, Ludwig Krämer (1996: 7), Head of the Waste Management Unit within the Environment Directorate of the Commission lamented that: 'There are only a few areas of Community law in which the difference between the written law and the practice is as great as in the case of Community environmental legislation.' The Commission's sixteenth report XVIth Report on monitoring the application of Community law noted that these problems still remain, chiefly as a result of, first, the difficulties some Member States experience in implementing EU law and, second, the Commission's limited right to monitor national compliance on the ground (CEC, 1999a). The Commission's most recent measures to improve policy implementation are discussed further in Chapter eight.

3.5 Conclusions

This chapter has sought to identify the key political processes affecting the implementation of European Union environmental policy. Three main themes have emerged. First, EU environmental policy has been transformed from its initially uncertain legal base to become a comprehensive programme of reform and regulation across Europe. At the forefront of this lie clear commitments to sustainable development, the process of environmental integration and the inclusion of a broad range of social and economic actors within the policy process. Against this, the EU's efforts to reverse environmental decline in Europe have enjoyed only partial success. Whilst this can be attributed to a number of factors, the EU's allegiance to trade and economic development must be seen as a major influence on the direction and success of its environmental programme. Although EU environmental policy has enjoyed considerable success, its philosophical foundations remain rooted in the notion of weak

sustainability. Whether this is a sufficient response to the environmental challenges faced by the EU Member States is an issue which will be explored throughout this thesis.

The second theme concerns the confederal and inter-governmental tensions within the complex and sometimes discordant European political system. William Wallace (1996: 451) even describes it, perhaps slightly harshly, as:

An incomplete political system: a 'quasi-state,' without the coherent articulation of interests and political preferences characteristic of a well-developed polity ... [Within this] ... different governments, with different traditions of statehood and different myths of national identity, choose different issues [to assert their national sovereignty], further complicating the management of Europe's multilateral and multi-level government.

Whether or not one totally accepts this indictment, it is apparent that the EU's political deliberations are punctuated by conflicts between competing interests and visions of European integration (Weale, 1999). The impact of this on material policy decisions is clearly expressed in the system of concurrent majorities and the bargaining outlook it engenders during policy formulation (Weale, 1996). Whilst decisions over EU intervention in domestic policies are managed by the subsidiarity principle, the formulation and acceptance of common initiatives has proven a battleground between the Union's environmental leader and laggard states and other assorted key actors. In policy implementation, the adoption of directives as the primary regulatory technique has facilitated the acceptance and implementation of EU legislation but it has also exacerbated the difficulties in achieving harmonised environmental standards across the Union.

The third main theme is the poor implementation of the EU environmental programme (CEC, 1998a). The number of infringement proceedings has risen apace in recent years (Demmke, 1997; Environmental Data Services (ENDS), 1998a), whilst the procedures to resolve them remain ponderous and only partially effective (Collins and Earnshaw, 1993). As with the doubts concerning subsidiarity and policy-making by concurrent majorities, the current enforcement process may be more oriented towards managing the nuances of EU politics than the effective resolution of pressing environmental

problems. Tsoukalis (1997: 276) encapsulates many of the afflictions of EU policy-making thus:

Some of the main characteristics of this *sui generis* political system are the slow and inefficient method of decision-making, which is still close to an inter-governmental type of negotiation with multiple layers; poor transparency and accountability of its institutions; an administrative structure which has serious difficulty in coping with the wide range of functions and the financial resources entrusted to it; a large 'implementation deficit' which results from the highly decentralised nature of the system and the difficulties experienced in exercising effective control, accompanied by the threat of sanctions, over the proper implementation of decisions made in Brussels; and perhaps more importantly, the lack of a popular base which goes hand in hand with the lack of democratic legitimation.

This is not to say there are perfect solutions to any of these issues; there are probably not. However, it is important to recognise that the mechanisms which are necessary to facilitate the functioning of the *corpus* EU - subsidiarity, consensual interest-led bargaining, decision-making by concurrent majorities and the push-pull dynamic - are not necessarily the same as those required to achieve an effective environmental programme. To sum up, the existing literature has discussed the philosophical, institutional and political foundations of EU environmental policy and identified several major obstacles to the construction of sustainable development within this complex political dynamic. Having completed this task, the thesis now considers the methods used to examine the expression of these issues and those raised in Chapter two during the implementation of the Packaging Waste Directive.

Chapter Four

Research Methodology

4.1 Introduction

Earlier chapters outlined a range of ideological, political and practical issues central to the implementation of EU environmental policy. As an introduction to the empirical element of this study, this chapter reviews the methods used to collect and analyse data. It re-caps on the aims of the study then outlines the overall research strategy adopted, its original contribution and the methods employed. In developing a methodological approach for this research, previous studies of environmental policy implementation, particularly from the field of waste management, were consulted. These included policy implementation studies by, amongst others, Michaelis (1995), Waite (1995) and Whiston and Glachant (1996), theoretical examinations of waste management policy (such as Pearce and Turner, 1992; Brisson, 1993), and qualitative analyses of policies and their implementation (Lister, 1996; Golub, 1996; Newton and Harte, 1997; Gibbs *et al.*, 1998; Eden, 1999). The information from these was supplemented by reference to specialist methodological guides (including Sarantakos, 1993; Czaja and Blair, 1996; May, 1997).

4.2 Research Strategy

4.2.1 Research Aims

The central aim of this thesis, identified in Chapter one, is to evaluate the extent to which price-based environmental regulation is capable of promoting the objectives of EU environmental policy. These objectives were identified in Chapters two and three as the advancement of sustainable development and the EU's environmental principles. Whilst both series of concepts are extremely broad and problematic to assess, their operationalisation in this study is explored in Section 4.4. The general approach adopted in relation to sustainable development is that of the spectrum of weak and strong sustainability proposed by Turner (1993), as this recognises the diverse interpretations expressed within the literature.

4.2.2 Research Themes

Essentially, the research assesses the influence of normative and positive factors on the outcome of EU environmental policies (see Chapter three). On the normative side, it seeks to determine whether price-based environmental policy instruments offer an effective means of achieving the sustainable management of packaging waste. Previous research has usually measured environmental policy success in terms of a combination of the environmental- and economic-efficiency factors reviewed in Chapter two (see also Bohm and Russell, 1985; Brisson, 1993; Michaelis, 1995). However, as environmental policy outcome is invariably affected by both normative and positive factors, this research also examines the relationship between EU decision-making structures and the effective operation of price-based environmental regulation.

In developing research questions, it is relatively easy to operationalise the first theme into measurable and testable hypotheses (there is/is not a significant relationship between price-based regulation and particular indicators of sustainable development). However, the second theme requires a more analytical and critical approach. Both qualitative and quantitative techniques were therefore used during the study. Whilst the main empirical analysis uses deductive and quantitative techniques, discussions on the relationship between EU policy-making structures and price-based regulation adopt a more inductive approach (see Holt-Jensen (1981), Johnston (1983), and Saunders *et al.* (1997) for summaries of the merits and limitations of inductive and deductive research). The move away from a rigid adherence to particular philosophical and methodological stances has gained increasing acceptance in human geography as researchers have recognised that no single technique can fully capture the meaning of the social world. Instead, 'multi-method' research is becoming an obvious choice in the conceptually diverse discipline of human geography (Philip, 1998). This study is therefore based on the general methodological approach suggested by McCall and Bobko (1990: 412):

What one's method reveals about the problem and how well one executes whatever method is chosen seems significantly more important [than rigid methodological stance].

The approach adopted during this research also follows the guidelines provided by Giddens (1993: 20):

Scientific work depends upon a mixture of boldly innovative thought and the careful marshalling of evidence to support or disconfirm hypotheses and theories. Information and insights accumulated through scientific study and debate are always to some degree *tentative* - open to being revised, or even completely discarded, in the light of new evidence or arguments (original emphasis).

It is nevertheless important to recognise that research can never be entirely value free because observations and interpretations are invariably influenced by the individual and received world-view of the researcher undertaking the investigation (Williams, 2000). Not only is total value freedom unattainable, it may also be undesirable if it denies the possibility of alternative perspectives on research problems - the criticism frequently levelled at logical positivism (Williams and May, 1996). This research essentially comes from an environmentalist perspective, in that the primary motivation is the development of knowledge that furthers the debate on implementing sustainable development. Thus, it is primarily concerned with the contribution made by the EU and industrial concerns to environmental sustainability rather than the processes themselves (see Zito (2000) as an example of an alternative perspective on EU environmental policy analysis). However, whilst it is important to recognise the existence of values, particularly in the social sciences, it is essential that academic research should strive for objective analysis throughout all stages of the research process (Weber, 1974).

4.3 Research Contribution

Numerous analytical texts have been written on the subject of environmental policy implementation. These generally take the form either of legal and policy analyses covering positive issues (Segerson, 1996; Lowe and Ward, 1998a; O'Riordan and Voisey, 1998), normative studies (Pearce and Turner, 1992; 1993; Brisson, 1993), or empirical research. Empirical examinations of environmental policy may be further divided into three general categories. The first is quantitative work exploring attitudes to environmental policies from a sociological perspective (Pelletier *et al.*, 1996; Grenstad and Wollebaek, 1998; Ebreo *et al.*, 1999). The second is qualitative studies of corporate environmental performance (Newton and Harte, 1997; Gibbs *et al.*, 1998; Eden, 1999). The third category is policy impact studies, often conducted on behalf of governments, which generally presents quantitative data without specifying the fieldwork methods used (Organisation for Economic Co-operation and Development (OECD), 1994; DTI/DoE, 1991; 1992). This latter category also rarely examines the

theoretical underpinnings of particular policy approaches and instead focuses entirely on implementation practicalities. Only a few works, notably Labatt (1991; 1997a; 1997b) discussing discretionary corporate responses to environmental initiatives, have attempted similar quantitative studies of business behaviour from a geographical perspective.

It is also obvious that these studies come from a variety of academic disciplines, each of which adds fresh dimensions to the understanding of environmental policy. However, human geography has two valuable roles to play in this research area. First, its strong empirical tradition can help in evaluating the practical usefulness of theories proposed by more academically abstract disciplines. Second, it is well equipped to analyse the spatial effects of environmental policies (Gibbs and Healey, 1997). Nijkamp (1980) argues strongly that though predictive modelling techniques can help inform the evaluation of environmental policies, ultimately these theories require empirical substantiation. Lélé (1991: 619) even brands such econometric modelling as 'arcane'. More moderately, Dixon (1990: 189) remarks:

Economists are increasingly being asked to show how their theories and techniques can be used to address real, immediate problems, both at the project and at the policy level. The record to date is mixed. In part this is a natural result of the inherent limitations of economics from a theoretical basis and the diversity of problems it is being called upon to address. As a science, economics is an empirical, quantitative discipline that is ill-suited to address certain subjective or qualitative topics. The 'value' of human life is a well-known example ... others abound in the environmental/natural resource management field.

Therefore, to re-cap from Chapter one, the main original contributions of this thesis are, first, its evaluation of the practical benefits of price-based environmental regulation and, second, its detailed examination of both the technical and the political determinants of environmental policy outcome. Its originality thus stems from its examination of the practical interaction between normative and positive factors during the formulation and implementation of EU environmental policy.

4.4 Research Methods

4.4.1 Introduction

Having reviewed the overall approach adopted for this study, the following section identifies and justifies the methods used to collect and analyse data. The research process consisted of five main stages:

1. An initial pilot study of the UK Packaging Waste Regulations prior to their implementation
2. Reviews of academic and professional document and literature sources
3. Qualitative interviews and correspondence with businesses affected by the Packaging Waste Directive
4. Quantitative and qualitative analysis of recycling infrastructure in Britain and Germany
5. A quantitative survey of business in Britain and Germany obligated to recover and recycle their packaging waste as a result of the Directive

However, one of the initial objectives in planning the empirical research was the selection of suitable case study areas in which to examine the implementation of the Directive. Considering the nature of the EU implementation process, the obvious choice was to examine the packaging waste policies of two Member States in order to compare the effects of different implementation strategies on policy outcome. Britain and Germany were selected for three main reasons. First, as two of the largest Member States, their policies are likely to have a significant impact on the production and management of packaging waste in the EU. This is especially true of Germany, as several other Member States have adopted variants of its packaging waste system. Second, since Germany's packaging legislation has been operating for significantly longer than Britain's (see Chapter five), temporal influences on policy outcome can be assessed (London and Llamas, 1994). Finally, the two countries have traditionally differed in the way they prioritise and implement environmental policies. Whilst it is always dangerous to indulge in cultural stereotyping, German environmental policy since the 1970s has largely been organised around *Vorsorgeprinzip* (the precautionary principle), strict legislation, and the promotion of high environmental standards (Zito, 2000). By contrast, Britain is usually characterised as being a reluctant or a pragmatic

participant in environmental policy (Lowe and Ward, 1998a; Zito, 2000), as wishing to base initiatives on scientific evidence rather than the precautionary principle, and as seeking to achieve cost-effective environmental protection (see Chapter three). The aim of selecting Britain and Germany as case studies, therefore, was to examine the extent to which these differences have affected the application of price-based regulation.

4.4.2 Pilot Study

The pilot research was undertaken as part of a Masters degree at the University of Plymouth and consisted of two elements; a case study of a business complying with the UK Packaging Regulations and a postal survey of 250 companies within the construction industry (Bailey, 1997). This stage of the research served two purposes. First, it explored the potential of the Packaging Waste Directive as an avenue for further research; second, it assisted in the development and testing of research methods for use in later studies. So as to avoid unnecessary duplication, the methods used in the pilot study are reviewed as part of the main research sections.

4.4.3 Documentary Research

In addition to the normal review of the academic literature, a range of documentary sources were used to develop a general understanding of the research area (Brannick, 1997). In particular, the aim was to explore the legislative frameworks used to implement the Packaging Directive in the Member States. Five main types of document were examined:

1. Existing academic analysis of packaging and recycling systems
2. Government and other official documents, including legislative instruments, parliamentary debates and administrative circulars
3. Consultants' reports, usually conducted on behalf of the European Commission or national governments
4. Industry and trade organisation documents
5. Specialist press reports

The existing literature contains numerous studies examining packaging waste policy (for example, Pearce and Turner, 1992; Waite, 1995; Michaelis, 1995; Whiston and

Glachant, 1996; Defeuilley and Godard, 1997; Bailey, 1999a; 1999b; 2000). Though few have attempted extensive empirical analysis of polluter responses to environmental regulation, they nonetheless provided useful perspectives on the process of policy implementation. For example, Pearce and Turner (1992) examine the potential of packaging taxes as a means of promoting the PPP, Michaelis (1995) and Defeuilley and Godard (1997) examine the economic efficiency of selected national packaging systems, whilst Whiston and Glachant (1996) and Fenton and Sinclair (1996) discuss 'voluntary' agreements between industry and government as frameworks for packaging stewardship schemes. Therefore, despite their varying methodologies and foci, these studies provided invaluable background on the nature of packaging waste management systems in the EU. Chapter five, which explores the nature of national packaging recycling systems within Europe, discusses these works in greater detail.

Government legislation and parliamentary debates at both EU and national level were also extensively reviewed in the early stages of the study (for example, *Hansard*, 1997; OJEC, 1993; *Debates of the European Parliament*, 1994). Legislative analysis encompassing both governmental levels is an essential pre-requisite of any research examining EU environmental policy because the Commission uses directives as its main legislative catalyst. As Chapter three highlighted, directives are only binding in terms of the obligation to act and the standards to be achieved (Lister, 1996). This means that the methods used by national authorities to achieve EU standards can vary considerably (Lowe and Ward, 1998a; Bailey, 1999a) and that detailed examination of EU and Member-State legislation is an essential part of understanding the implementation process. The legislation examined is shown in Table 4.1.

The British government also produced numerous consultation papers concerning the UK Packaging Regulations (DoE, 1996b; Department of the Environment, Transport and the Regions (DETR), 1998a; 1999a; 1999b). These again provided important detail on the methods used to implement the Directive in Britain. Corresponding information for Germany and other EU Member States was largely provided by specialist reporting agencies (Hagengut, 1997; Perchards, 1998) and through contacts with the UK Environment Agency, the German packaging organisation, *Duales System Deutschland* (DSD) and the *Arbeitsgruppe Umweltstatistik* (ARGUS) at the Technical University of Berlin.

Table 4.1 EU Packaging Legislation reviewed for the Research

<i>Jurisdiction</i>	<i>Legislation</i>
EU	The Packaging and Packaging Waste Directive (94/62/EC) (OJEC No L365, 31.12.94).
United Kingdom	The Producer Responsibility Obligations (Packaging Waste) Regulations 1997 (DoE, 1997) The Producer Responsibility (Essential Requirements) Regulations 1998 (DETR, 1998b) The Producer Responsibility Obligations (Packaging Waste) (Amendment) Regulations 1999 (DETR, 1999c)
Germany	<i>Verordnung über die Vermeidung von Verpackungsabfällen</i> (Ordinance on the Avoidance of Packaging Waste) (<i>Verpackungsverordnung - Packaging Ordinance</i>), 12 June 1991 <i>Verordnung über die Vermeidung und Verwertung von Verpackungsabfällen</i> (Ordinance on the Avoidance and Utilisation of Packaging Waste) (BGB1. I 1998 S. 1998)

Though official documents are the starting point of most policy analyses, it was important to canvass the views of all parties affected by the Directive, including those who opposed the policy. As official documents stressed the active role taken by industry during the formulation and implementation of the British and German legislation (DoE, 1996b; DETR, 1998b), several industry groups were asked to supply copies of their responses to government consultations. In order to obtain a representative cross-section of opinions, organisations from each sector of the UK packaging chain were contacted, including industry and materials organisations, compliance schemes and reprocessing companies (see Chapter five)¹. Further details on industry's view of the Directive were gathered from specialist environmental and industry press service reports, such as *Environmental Data Services* (ENDS), *Materials Recycling Week* (MRW), *Raymond Communications* and *Packaging News*. Kiecolt and Nathan (1985), in particular, stress the value of secondary material as a cost-effective

¹ For opinions concerning legislation in EU states other than the UK, the European Organisation for Packaging and the Environment (EUROPEN), the Industry Council for Packaging and the Environment (INCPEN), the *Duales System Deutschland* (DSD), the *Arbeitsgruppe Umweltstatistik* (ARGUS) and Perchards were contacted.

and authoritative source of background material. However, they also urge caution in the interpretation of secondary data, as the apparent authority of published work can make it difficult to distinguish errors and value statements from 'reliable' facts. This point was particularly relevant to this research, as many documents were provided by businesses with a vested interest in influencing the debate on the Packaging Directive. Nonetheless, their use within this study enabled a number of major themes to be quickly and comprehensively reviewed.

4.4.4 Preliminary Qualitative Research

As the original pilot study was conducted prior to the implementation of the UK Packaging Regulations in 1997, it focused principally on how affected companies *believed* they would be affected by the legislation prior to its format being officially finalised. The subsequent document search revealed that the mechanisms used to implement the Directive in Britain had altered radically between the initial and main studies². Further exploratory data collection was therefore necessary to understand how these changes might influence business responses to packaging legislation. Whilst information on the main changes had been provided by documentary data (such as Perchards, 1998), a selection of businesses was contacted in order to discuss the mechanics of the UK Regulations. The main task in relation to the German legislation was to identify the nature of the amendments introduced since the 1991 Packaging Ordinance (see Chapter five). This was achieved through contacts with a variety of specialist organisations, including DSD, ARGUS, ENDS and Perchards³. However, language difficulties prevented extensive contact with organisations that did not provide information in English.

In total, 25 UK businesses were asked to provide details of their methods for complying with, and their opinions on, Britain's Packaging Regulations (see Appendix 1 for contact letter). UK Respondents were selected at random from an Environment Agency database of companies obligated under the Regulations (see Ackroyd and Hughes,

² For example, the introduction of the Packaging Waste Recovery Note (PRN) system (see Chapter five), the main mechanism for proving compliance with the Directive's recovery and recycling targets, was not discussed in detail in the original consultation documents in 1996 (DoE, 1996b).

³ Contacts were through the DSD email information line and its web-site www.gruener-punkt.de.htm. ARGUS is currently conducting a major review of packaging waste management systems in the EU on behalf of the European Commission.

1992; Maisel and Persell, 1996 for discussions of sampling techniques). Of these, 16 either provided standard corporate literature, responded directly to the questions raised, or made contact by telephone. Because written responses were requested, these calls were unexpected and could not be tape-recorded. Interview notes were therefore transcribed immediately following the telephone conversations. Whilst standard questions were used in the original letter, the responses in both letter and interview form were generally of an unstructured nature. This was not a significant problem, however, as this stage of research was primarily aimed at gaining exploratory information rather than a standardised dataset.

Though the qualitative interviews aided the generation of research questions and hypotheses, it is recognised that this research technique is susceptible to respondent bias. Robson (1993) and Judd *et al.* (1991) observe that respondents with strong opinions on a particular subject - particularly negative ones - are more likely to respond to surveys than those who are generally content or indifferent. However, because the intention was always to test the preliminary findings on a larger and more representative sample (Morris, 1993), the basic objective of this research stage was adequately achieved.

4.4.5 The Development of Recycling Infrastructure in Britain and Germany

The review of documentary sources and academic literature revealed, amongst other things, that packaging waste management systems should possess several key attributes in order to achieve their policy aims effectively. These include the establishment of balanced reprocessing and waste collection infrastructure, the participation of industry and public actors, the development of an effective financing mechanism, and the co-ordination of each key sector of the recycling industry (Brisson, 1993; Michaelis, 1995; Waite, 1995; *Hansard*, 1997; Turner *et al.*, 1998). As part of the assessment, therefore, it was necessary to examine how each of these functions was being managed in Britain and Germany. In Germany, this could largely be achieved through secondary sources and contacts with recycling organisations. In the UK, it was considered necessary to supplement secondary information with a postal survey of recycling (reprocessing) companies.

The sampling frame for this survey was a database of business registered with the Environment Agency as *accredited reprocessors* of packaging waste⁴. It was decided that a census survey would be the most effective approach as the database contained only 133 independent businesses. Notwithstanding the unlikely event of a 100% response, it was adjudged that sampling from within this frame would produce inadequate data. The technique of contacting all members of a particular population with the expectation of only receiving a sample of replies is generally termed an *incomplete census* (Jancowicz, 1991; Moser and Kalton, 1971: 54). All businesses on the register were contacted by letter. This stated the aims of the research, provided assurances of confidentiality, and asked respondents' opinions on key aspects of the UK Regulations (Appendix 2). If no reply was received within three weeks, the company was contacted again. However, because few replies were received in response to the follow-up letter, no further contacts were attempted.

The survey combined a mixture of quantitative and qualitative questions. The quantitative questions attempted to measure the performance of each reprocessing sector against the capacity requirements needed to comply with the EU Packaging Directive, whilst the qualitative questions sought to establish attitudes towards the financing and policing mechanisms introduced by the government. Because the response rate was lower than hoped (36.1%), both quantitative and qualitative data could be examined manually rather than using specialist software. Responses to qualitative questions were graded into positive, neutral or negative comments on particular questions. They were also divided according to each reprocessing sector affected by the Packaging Regulations (paper, plastics, glass, steel, aluminium and wood) (DoE, 1997), the aim here being to assess differences of opinion within and between individual materials sectors. However, the size of the data set meant that it was not possible to use statistical analysis techniques. Shortly after this survey, the Department of the Environment, Transport and the Regions (DETR) undertook a similar review, which included data from all accredited reprocessors (DETR, 1999a). Whilst this obviated the need for the quantitative data yielded by the survey, the information available for analysis was significantly improved.

⁴ Used with permission and acknowledgements.

4.4.6 Survey of Firms affected by Packaging Legislation in Britain and Germany

The main empirical component of the study was the survey of businesses affected by packaging legislation in the UK and Germany (the *Packaging Producer Survey*). Its overall aims were; first, to identify the waste management techniques used by packaging producers in the two countries in response to their respective legislation and, second, to assess the extent to which price-based environmental regulation had influenced their actions. The postal survey technique was chosen because it provided more extensive coverage of business responses than could be achieved using personal or telephone interviews (see Table 4.2 for a review of the relative strengths and weaknesses of quantitative and qualitative research methods)⁵. At the time the survey was conducted, nearly 4,000 UK business were obligated by the UK Regulations (DETR, 1998b) - this figure is set to rise to approximately 11,000 in the year 2000 (DETR, 1999d) - whilst over 17,000 German companies have some form of recycling responsibilities as a result of the Ordinance (DSD, 1998). While interviews may have revealed more in-depth information about the compliance methods used by a small selection of companies, these results would have been difficult to generalise reliably. Therefore, despite the shortcomings of postal surveys (lack of detailed information, difficulties in gaining respondent validation, and the potential for misinterpretation) (Moser and Kalton, 1971; Czaja and Blair, 1996), quantitative analysis was considered the most appropriate method for achieving the study's main aims.

Undoubtedly the biggest drawback of self-administered postal survey, however, is the fact that the technique is typified by low response rates. Morris (1993) claims that a 15% response rate is quite common. However, response rates can be substantially improved through careful research and questionnaire design; furthermore, many problems had already been identified and rectified during the pilot study. Moser and Kalton (1971) and Czaja and Blair (1996) highlight a number of research design factors which need to be considered in any postal questionnaire; sampling technique, questionnaire design, piloting and refining the survey, survey administration, and data analysis. The remainder of the section details the methods used in each stage of the survey.

⁵ For an extensive review of interviews and questionnaires in social research, see Judd *et al.* (1991) and Sarantakos (1993).

Table 4.2 Quantitative and Qualitative Approaches to Social Research

	<i>Quantitative</i>	<i>Qualitative</i>
Advantages	<p>Economical collection of large amounts of data</p> <p>Clear theoretical focus for the research from the outset</p> <p>Greater opportunity for researcher to retain control of research process</p> <p>Easily comparable data</p>	<p>Facilitates understanding of how and why</p> <p>Enables researcher to be alive to changes which occur during the research process</p> <p>Good at understanding social processes</p>
Disadvantages	<p>Inflexible – direction often cannot be changed once data collection has started</p> <p>Weak at understanding social processes</p> <p>Often does not discover the meanings people attach to social phenomena</p>	<p>Data collection can be time consuming</p> <p>Data analysis is difficult</p> <p>Researcher has to live with uncertainty that clear patterns may not emerge</p> <p>Generally perceived as less credible by 'non-researchers'</p>

Source: adapted from Saunders *et al.* (1997: 74)

Sampling

The first task in the survey process was to obtain databases of businesses affected by national packaging legislation in each country. Whilst this was quite straightforward for Britain, in that the Environment Agency maintains a public register of obligated companies, no corresponding database exists for Germany in the public domain. Although numerous organisations were contacted, including the German Environment Ministry⁶, the *Umweltbundesamt* (German Environment Agency), the DSD and the *Gesellschaft für Verpackungsmarktforschung* (GVM - German organisation for packaging market research), it was ultimately necessary to resort to a commercially-produced directory of German companies. The difficulty with this was that the classifications used in the directory (for example, electrical goods, food products) did not match those employed by the Packaging Ordinance (see Chapter five).

⁶ *Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit* (BMU).

In order to produce a German sampling frame that was representative of businesses affected by the Ordinance and which could be compared with the UK dataset, two selection criteria were used. First, the main industry sectors targeted by the Ordinance - manufacturers, distributors and retailers - were included (see Chapter five, also Hagengut, 1997; Perchards, 1998). Second, companies were selected on the basis of company turnover. The reason for this was that, at the time, the UK Regulations exempted businesses with a turnover of less than £5 million or handling less than 50 tonnes of packaging a year from direct recycling responsibilities. Although the German Ordinance contains no equivalent provision, comparing companies of significantly different size and functional characteristics would clearly have prejudiced the analysis. Therefore, all German businesses with an annual turnover of less than £5 million were excluded. As neither data set contained information on packaging consumption, this factor was ignored. Using this method, a database of over 4,500 companies was created. The exclusion of smaller businesses and, inevitably, the limitations of the electronic directory explain the discrepancy between this and the total number of companies obligated by the Packaging Ordinance. Although this strategy was not ideal, the sample derived was sufficiently large that the chances of bias were minimised as much as possible (Maisel and Persell, 1996).

The next stage was to draw a sample of 900 companies from each frame, based on the guideline that statistical analyses usually need 600 or more respondents to be reliably generalisable (Babbie, 1989). In both cases, the sample was drawn using random numbers generated on Excel spreadsheets. However, the German sample was also stratified according to the representation of each activity sector in the original business directory. The technique of stratified sampling is commonly used within social research to ensure that the respondent group accurately reflects the composition of the overall population set (Oppenheim, 1992; de Vaus, 1996). This was important in this case because of the difficulties in obtaining a wholly reliable database of German companies. The fear was that, despite extensive analysis of the 1991 and 1998 Ordinances, the sample would be dominated by a sector whose involvement in recycling was minimal. The best way to minimise this risk, therefore, was to stratify the sample. However, this technique was not ideal as the UK database contained no details of company characteristics and, therefore, similar stratification could not be repeated. Against this, it was known that, notwithstanding a few inaccuracies, the

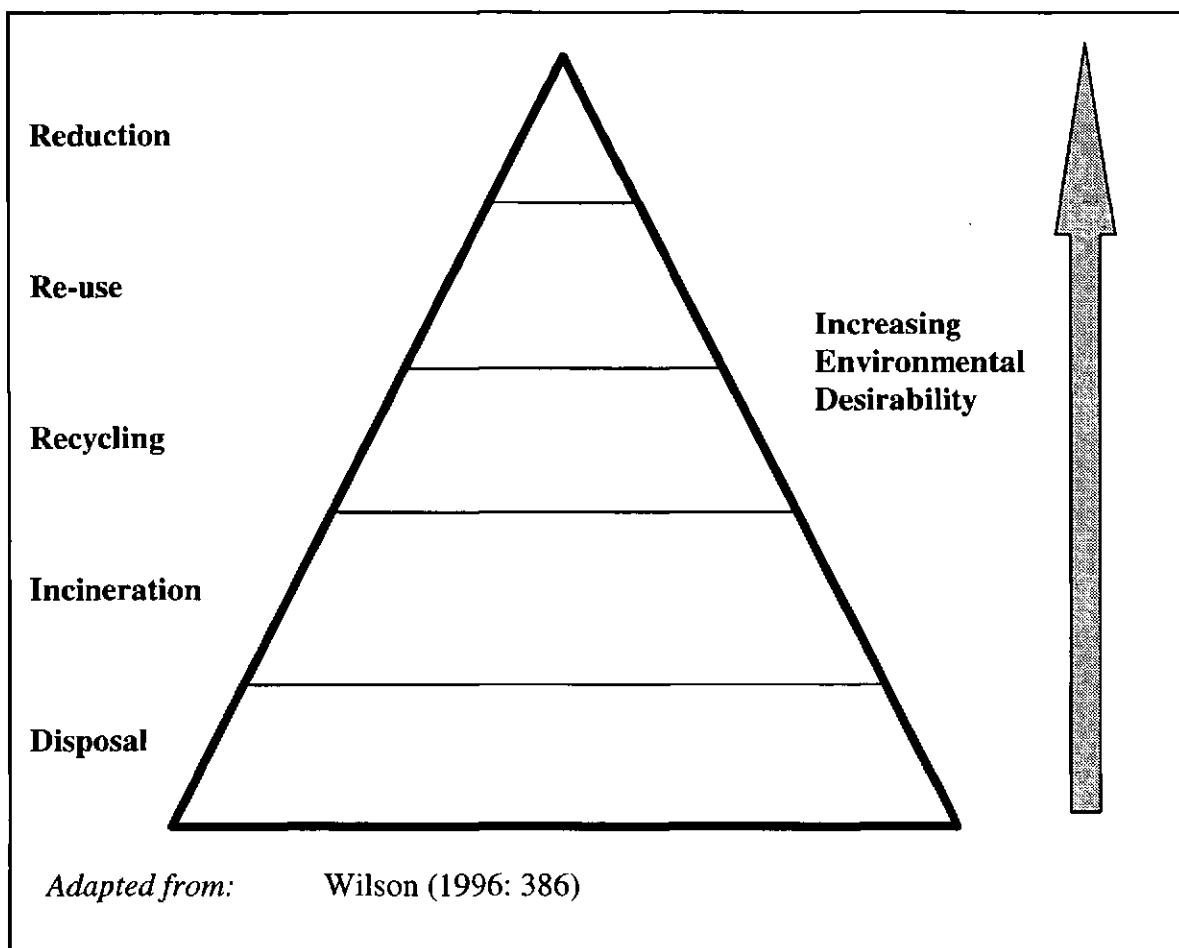
Environment Agency register was representative of the desired population set. However, it was not possible to say with absolute conviction that the two final samples were entirely equivalent, though strenuous efforts were made to remove significant differences. The implications of these problems are assessed further in the first results chapter (see Chapter six).

Questionnaire Design

DeVellis (1991) identifies two critical stages in questionnaire design, the operationalisation of research aims into specific themes using general theory and specific context as guides to question content and order, and the generation of a pool of questions to transform themes into measurable variables. The difficulty highlighted in Chapters two and three, however, is the operationalisation of such concepts as sustainable development and EU environmental principles into readily measurable variables (Redclift, 1987; Lister, 1996). The translation of general themes into specific contexts is a constant problem for environmental policy studies and one usually only overcome by focusing on limited aspects of the wider concepts (Trudgill and Richards, 1997). Pearce and Turner (1992), for example, use waste minimisation as a single assessment criterion, and Brisson (1993) focuses almost exclusively on the economic efficiency of waste management systems. However, because the main method of measuring the success of environmental policies, compliance with legislative standards, has severe limitations, this study sought to explore evaluation methods that took account of implementation methodology and a broader range of environmental performance criteria. It was therefore decided to base the study's empirical evaluation primarily around the Waste Management Hierarchy (WMH). The hierarchy has been used by numerous waste management studies (for example, Allaway, 1992; Levenson, 1993; Fenton and Hanley, 1995; Read *et al.*, 1998). In simple terms, it ranks methods of waste management according to their environmental impact (Wilson, 1996; DETR, 1999b) (see Figure 4.1). At the top of the hierarchy as the least environmentally deleterious option is source reduction; this is followed in turn by re-use, recycling, incineration with energy recovery and, finally, landfill disposal. The waste hierarchy was particularly appropriate for this study as it encapsulates both the legislative standards adopted in the Directive (recycling and incineration) and its general 'essential objectives' (the promotion of waste prevention and re-use) (see Chapter five).

Furthermore, the WMH has been used as a general framework for EU waste policy since the Second Environmental Action Programme (EAP) in 1977 (Read *et al.*, 1998).

Figure 4.1 The Waste Management Hierarchy



Strict interpretations of the waste hierarchy are not universally accepted, however. Collins (1996), for example, notes that the environmental rationale for paper recycling is particularly suspect compared with that for incineration. Similarly, Barrett and Lawlor (1997) argue that the hierarchy is spatially insensitive, in that the impact of recycling in rural areas (where longer journeys to recycling centres are required) is greater than that of landfill. Finally, within the specific context of the Packaging Directive, Golub (1996) notes that the WMH was not officially adopted in the legislation, though most elements were included in its objectives. Consequently the hierarchy should be seen as a qualitative guide to the environmental desirability of waste management options rather than as a prescriptive order of merit.

In addition to employing the WMH as a framework for evaluating business reactions to the Directive, the financial impact of price-based regulation on respondent businesses was explored in the questionnaire. The full list of themes explored was therefore the following:

- Business compliance with the Directive's legal standards and 'essential objectives' between 1997 and 2001
- The operational and financial impact of national legislation on affected companies
- Corporate opinions on the format and efficacy of packaging legislation, including the effectiveness of price-based regulation and alternative policy instruments.

Besides the main research themes, respondents were also asked to provide basic company-profile data, including turnover, employees, sector of the packaging chain and main business activity. These measures enabled the influence of business profile on other indicators and the comparability of the samples to be monitored. Alderman and Fischer (1992) and Labatt (1997a) both favour the use of employment figures as a measure of company size but because the British and German legislation differentiate businesses primarily on the basis of activity sector (and, in Britain, financial turnover), financial measures of corporate activity were also included. Copies of the final questionnaires are provided in Appendices 3a and 3b.

The second stage of questionnaire design is the generation of questions to transform research themes into measurable variables (DeVellis, 1991). The first decision was the depth of information required from respondents and, therefore, the balance between open and closed questions. Most methodological guides suggest that open questions are time-consuming and difficult to interpret in large-scale surveys (Gill and Johnson, 1991; de Vaus, 1996)⁷, a problem that was compounded in this research by the need to translate any non-numerical answers provided by German respondents. Categorising and comparing responses to open questions can also be problematic and subjective. Therefore, in order to maintain generalisable, comparable and accessible data, most questions were closed and pre-coded even though this meant some issues could only be

⁷ Though several well-recognised software packages exist for qualitative data analysis, for example, NUDIST (Non-numerical Unstructured Data Indexing Sorting and Theorising).

explored relatively superficially. In terms of question content, Converse and Presser (1986) stress the need for simplicity, intelligibility and clarity. Though this appears an obvious statement - notwithstanding the fact that most questions were re-phrased several times in order to improve their clarity and focus - this study generally targeted managers with detailed knowledge of packaging legislation and how it related to their companies. The survey was therefore able to raise more complex issues than is usually feasible during postal surveys.

The next issue in questionnaire design is that of measurement. Dillman (1978) identifies four distinct types of measurable data, attitudes, beliefs, behaviour and attributes. A mixture of nominal and interval measurements was used to quantify two of the questionnaire's themes, measures of legal compliance and actions to promote the WMH. Nominal measures were used to ascertain whether businesses were, for example, engaged in particular waste management activities, and interval data to quantify the extent of their involvement. Examples of the measurement types used are shown in Figure 4.2. However, though the collection of interval data provides readily measurable and comparable data, two difficulties must also be recognised. First, many companies are reluctant to provide commercially sensitive information about their businesses; second, data may not be measured by the organisation in the format requested (Labatt, 1997a). Either eventuality may discourage businesses from replying to the questionnaire. It is therefore often preferable to request sensitive information in ordinal format in order to provide an additional element of respondent confidentiality (Robson, 1993). This tactic was used, for example, in relation to the turnover and number of people employed by respondent companies.

Although the study deliberately did not engage in detailed qualitative research, it was felt that business' attitudes towards the policies used to implement the Packaging Directive should nonetheless be canvassed. In order to achieve this, two series of proposition sets were used to explore (i) whether businesses felt national policies were promoting greater environmental stewardship (Section C of the questionnaire), and (ii) their opinions on the general policy format (Section D).

Figure 4.2 Nominal and Interval Data Measurements used in Producer Survey

Example 1: Nominal Data

A1 Is your company aware of the Producer Responsibility Packaging Waste Regulations? **IF NO, ANSWER SECTION E ONLY^a** 1. Yes 2. No

Example 2: Interval Data

B1 Approximately what percentage (by weight) of packaging was reused in 1997?

B2 Approximately what percentage (by weight) will be reused by the year 2001 ?^b

Notes

a Similar instructions and prompts were used throughout the questionnaire to guide respondents, a technique recommended by Moser and Kalton (1971) and Robson (1993).

b Example 2 demonstrates the importance of defining the required information specifically and carefully to minimise respondent confusion (Robson, 1993).

The sets were measured using Likert attitudinal scales (see DeVellis, 1991 and Oppenheim, 1992 for detailed discussions of Likert scales), a technique widely used in social and geographical research. Their first strength is that they produce ordinal data on beliefs and attitudes that can be analysed statistically. In addition, they are familiar to many respondent groups because of their extensive use in market research surveys. However, Likert scales also have several limitations. First, they provide superficial information compared to in-depth interviews (see Gibbs *et al.*, 1998 as an example of interviews in environmental policy analysis). Moreover, it is often difficult to compare attitudinal data produced by studies where different scaling systems are used (de Vaus, 1996). Standardised five-point Likert scales were used for all the attitude measures employed in the survey (see Figure 4.3 for examples). Social research texts usually recommend that between five- and seven-point Likert scales are used to provide sufficient sensitivity without confusing respondents or creating spurious levels of detail (DeVellis, 1991; Sarantakos, 1993).

Figure 4.3 Likert Scale Questions used in the Producer Survey

<i>Specific Company Actions</i>	+2 strongly agree	+1 agree	0 neither	-1 disagree	-2 strongly disagree
The company will only be able to comply with the minimum standards set by the Regulations	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
<i>Company Attitudes</i>	+2 strongly agree	+1 agree	0 neither	-1 disagree	-2 strongly disagree
The Packaging Regulations will achieve a cost-effective solution to Britain's packaging waste problems	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>

Piloting and Refining the Questionnaire

A series of piloting stages were undertaken before the survey was administered, though the Masters research had already established that the proposed questionnaire layout generally worked well with corporate respondents. First, the questionnaire was reviewed by peers and supervisors at the University. Second, it was sent to two external academics working in similar research fields. Finally, questionnaires were sent to 20 businesses in each country. Each questionnaire was accompanied by a cover letter explaining the research and a feedback form using a framework of key points suggested by Saunders *et al.* (1997) (see Appendix 4). The most frequent problems were questionnaire length - the original was five A4 sides - and ambiguous questions. Though the majority of responses were generally favourable, two UK businesses criticised certain aspects of the questionnaire. It was therefore decided to re-submit the revised questionnaire to 20 further companies to ensure all questions were relevant and clear. The piloting process also enabled tentative response rates for the survey to be estimated (Table 4.3).

Table 4.3 Predicted and Actual Response Rates from Producer Survey

	Pilot Study <i>% response</i>	Main Survey				
		<i>Posted</i>	<i>Predicted Response Rate</i>		<i>Actual Response Rate</i>	
			<i>Returned</i>	<i>%</i>	<i>Returned</i>	<i>%</i>
UK	70.4	900	360	40.0	469	52.1
Germany	n/a	900	225	25.0	309	34.3
TOTAL	70.4	1800	585	32.5	778	43.2

Administering the Survey

The first issue to consider in conducting the survey was its timing. Previous experience suggested that the latter part of the year should be avoided because many companies would be too busy to complete unsolicited questionnaires (see also Jancowicz, 1991). It was therefore decided to administer the survey between January and March, as this is generally the quietest time of the year for manufacturing and retail companies. The second issue was to ensure the questionnaire was received by the most appropriate member of staff in respondent companies. Labatt (1997a) suggests that telephoning businesses to obtain contact names can significantly boost response rates. However, the size of the sample and language difficulties made this exercise unfeasible. Furthermore, neither the Environment Agency database nor the original German business directory contained reliable contact information.

Instead, a European business directory was used to obtain names for the German respondent group, though this was not particularly successful as less than 250 of the 900 businesses were listed on this particular register. Also, this approach may be counter-productive if the directory is out of date, as questionnaires sent to ex-employees are more likely to be ignored. It was therefore decided not to repeat the process for the UK but, instead, to address the letter to 'The Managing Director' in the anticipation that it would be forwarded to the appropriate member of staff. Whilst this was a calculated risk, the final response rate appears to have vindicated the tactic.

All questionnaires were allocated a unique and confidential identification code before being posted. This enabled questionnaires to be monitored as they were returned whilst maintaining respondent anonymity outside the research (Oppenheim, 1992). Each questionnaire was accompanied by a cover letter (Appendices 5a and 5b), explaining the aims of the research, its contribution to understanding packaging policy, and guaranteeing respondent anonymity, confidentiality and protection from harm. Whilst the letter did not promise individual feedback, it indicated that the results may be published by the Institute of Wastes Management (IWM) (Bailey, 1999c). Pre-paid reply or International Business Reply Service envelopes were included with each questionnaire pack.

As replies were received, they were recorded against the database of questionnaire codes. All businesses not replying within three weeks were sent reminder letters (Appendices 6a and 6b) with further copies of the questionnaire and reply envelopes. Some researchers recommend that up to three reminder stages should be attempted; the first a letter, the second enclosing a copy of the questionnaire and the third containing an abbreviated form of the questionnaire (Adams and Schvaneveldt, 1991; Robson, 1993). However, the pilot study employed two reminder stages - a letter and a duplicate questionnaire. It was found that few businesses responded to the first reminder but that the second was more successful. This was probably because the original questionnaire had been discarded even if respondents did wish to reply. Because of this and resource constraints, only one follow-up stage was attempted. Although further reminders may have increased the number of responses slightly, the final rates achieved, 43.2% overall (52.1% for the UK, 34.3% for Germany), are generally above those expected from this data collection technique.

Data Analysis

A number of research texts discuss the subject of questionnaire coding (see Moser and Kalton, 1971; Czaja and Blair, 1996 and Parfitt, 1997). In accordance with the technique recommended by de Vaus (1996) for streamlining closed-question surveys, all questions were pre-coded before the questionnaires were sent out. The main preliminary tasks were therefore to check the accuracy of data punching and to ensure that only relevant businesses were included in the data analysis (Parfitt, 1997). Two

filter questions were used to identify companies for whom the research did not apply (see Table 4.4). This analysis also enabled the accuracy of the sampling frames to be assessed; further details of this analysis are provided in Chapter six. Completed questionnaires were input onto SPSS data sheets for analysis of business characteristics within each country and comparison of the two respondent groups. The purpose of the data analysis was then to determine, first, the waste management actions employed by respondent businesses, second, the link between actions and environmental taxes and, third, the attitudes of businesses in the two countries towards packaging waste legislation.

Table 4.4 Responses to Filter Questions

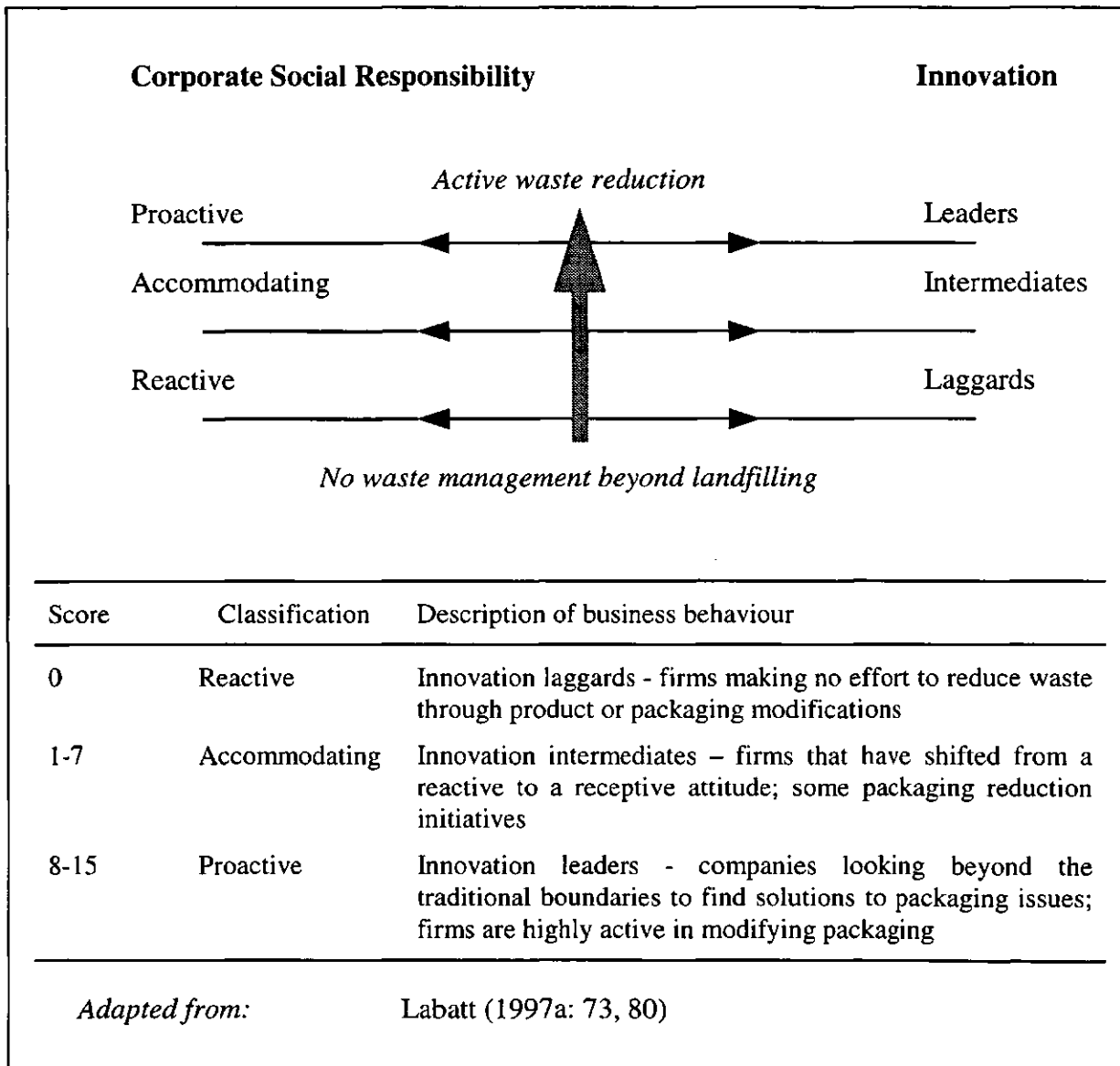
	Number of respondents	% of companies aware of national packaging legislation	% of companies with legal responsibility under national packaging legislation
UK	469	98.7	95.9
Germany	309	93.2	76.4
Total	778	96.5	88.2

A standardised sequence of statistical procedures was used to examine the data. Frequency and descriptive statistics were produced for each variable, along with histograms for all ordinal and interval measures. The aim of this was to assess the distribution of the data and the applicability of parametric tests (Clegg, 1990). This revealed that very few ordinal or interval measures were normally distributed. The next stage was to assess the impact of company characteristics on the data, using Chi-Square (χ^2) for nominal data and Kruskal-Wallis one-way Anova for ordinal and interval data (Ebdon, 1985).

Following this, the variance of each attitude indicator from point zero (neutral attitudes) was tested using one-sample t-tests. Although the one sample t-test could be used for this purpose because it is reasonably robust to non-normal distribution, the non-parametric Mann-Whitney tests were used to compare the two samples (Shaw and Wheeler, 1994). Similarly, Spearman's rank was used to measure correlation between

ordinal and interval variables. A range of multi-variate techniques, including multiple regression and factor analysis, were also used, though these revealed little by the way of significant causal relationships. Further details of this analysis are given in the study's results in Chapter six.

Figure 4.4 Classifications of Corporate Environmental Response



Labatt's studies (1991; 1997a) of corporate waste management in Canada used similar variables as those employed in this research to construct a series of corporate waste minimisation indices. Labatt (1997a) identified three levels of waste management,

reduction, re-use and recycling, then scored business environmental responses according to which initiatives had been adopted for their top five products (one point per product per initiative with a maximum score of 15 for each business). She then used the index to classify companies as either environmentally reactive, accommodating or proactive, based on the strength of their corporate social responsibility and innovation in response to environmental pressures (see Figure 4.4). Other authors have attempted to categorise corporate environmental response along similar continua⁸. Whilst the majority of these classifications are qualitative in nature, the development of aggregate waste management measures is one means by which corporate environmental response can be assessed.

Three similar environmental-response indices were created for this research:

a) *The Waste Management Hierarchy Actions Index (WMHA)*

For each waste management action undertaken by respondent companies (the collection of packaging waste at company sites or from customers, the reduction, re-use and purchase of recycled packaging), a score of five points was allocated. A business could therefore score a maximum of 25 points. The equation for this index is:

$$WMHA\ Index = \sum (\text{Waste Management Hierarchy Actions}) \times 5$$

This index was then refined to take into account the number of packaging materials covered by each waste management action (the $WMHA_j$ index). By adding a further point to the $WMHA$ score for each material, a business reducing consumption of all six materials would score five points for the action and six points for the materials included. However, the maximum score a business could achieve for the $WMHA_j$ index was 49 points, as materials collected from customers and own sites were only counted once⁹. The equation for $WMHA_j$ therefore becomes:

⁸ Bhargava and Welford (1996) classify corporate responses to environmental issues along a continuum they term ROAST (Resist, Observe, Accommodate, Seize and pre-empt and Transcend).

⁹ Maximum score = 5 x 5 points for waste management actions + 4 x 6 points = 24 for materials covered.

$$WMHA_j \text{ Index} = \sum (\text{Waste Management Hierarchy Actions}) \times 5 + \sum (\text{Materials included for each waste management action})^{10}.$$

b) *The Waste Management Hierarchy Targets Index (WMHT)*

The weightings given under *WMHA* and *WMHA_j* might be considered slightly arbitrary, in that all actions and materials are given the same score, regardless of their environmental impact. This was partially overcome by constructing a further measure based on the waste management targets set. *WMHT* therefore measures the combined targets set for the above actions, based on the equation:

$$WMHT \text{ Index} = \sum ((\text{collection \%} + \text{reduction \%}) + (\text{re-use \% 1997} + \text{re-use \% 2001}) \div 2 + (\text{purchases \% 1997} + \text{purchases \% 2001}) \div 2)$$

In this formula, the average re-use and recycled purchases figures for 1997 and 2001 were used so as to ensure an even representation for each waste management action. Consideration was also given to placing weightings that reflected the environmental desirability of each waste hierarchy action. However, it was decided that such a scaling would be too arbitrary, especially considering the contested nature of the waste management hierarchy.

c) *The Cost-burden v. Action Index (CbvA)*

This index was used to measure the correlation between the percentage waste management targets set by companies and the costs incurred complying with national packaging legislation. However, it was not possible to correlate percentage targets against the total costs sustained by companies, as larger companies would obviously incur higher overall liabilities because of their greater turnover and use of packaging. A more representative measure, therefore, was the proportionate compliance cost burden. This was calculated by dividing total costs by either company turnover or number of

¹⁰ The relative weightings given under the second index (five for a WMH action, one for each material) are designed to avoid it being unfairly weighted in favour of businesses which use a wide range of packaging materials, whilst still reflecting attempts to include a greater number of packaging materials within WMH actions.

employees to produce two *CbvA* indices. Thus, the correlation equation for *CbvA* turnover was formulated as:

$$\text{CbvA Index (Reduction)} = \text{Reduction \% 1997-2001 correlated against (compliance costs + company turnover (mid-point of category))}$$

Though these scales use similar methods as those developed by Labatt (1997a), it must be recognised that the weightings, particularly for the *WMHA_J* Index, involve elements of subjective judgement (de Vaus, 1996). These indices nonetheless provide aggregated measures of business response directly related to the main aims of the Directive. Those that involve such weightings, however, should be taken as comparative guides to corporate environmental response rather than as authoritative hierarchies. These indices were therefore devised as an experimental method for assessing business reactions to environmental policies containing multiple objectives.

4.5 Conclusion

The purpose of this chapter has been to review the methods used to conduct this research. In pursuit of the study's aims, the task was to establish a methodological approach that evaluated the implementation of EU environmental policy in the Member States *and* business responses to price-based regulation. Britain and Germany were chosen as the focus of study because of their significant influence over the implementation of EU policies and their contrasting approaches to environmental issues. The overall strategy was based around the notion that the research problem, rather than any pre-determined methodological stance, defines the methods employed. Whilst secondary data were used to examine the formulation and transposition of EU policies, both qualitative and quantitative methods were employed to evaluate their implementation. Qualitative techniques, including interviews and documentary sources, were used to provide background information, and quantitative methods in the form of postal surveys were used to assess business responses to packaging waste legislation and price-based regulation.

As with any research, it is recognised that there is no single 'correct' research methodology and that alternative techniques could have proven equally valid. As Ackroyd and Hughes (1992: 10) argue:

It is not clear that any one method has any intrinsic or canonical superiority over any other. All have their problems and are in the same boat as far as their worth and merits are concerned.

Bulmer (1988) also notes that there is no 'best' method for conducting research within organisations, but that successful studies need to work within the resources available and further practical and theoretical understanding equally. In particular, there is considerable scope for the use of qualitative interviews to explore the reasoning behind corporate responses to environmental policies. Such information would undoubtedly be of value to environmental policy-makers. Therefore, having justified the methods employed in this study, the next chapter examines the negotiation and implementation of the Packaging Waste Directive.

Chapter Five

The Implementation of the Packaging Waste Directive

5.1 Introduction

In simple terms, all policy-making processes can be divided into two basic components, the determination of desired objectives and the selection of policy instruments to achieve these aims (Segerson, 1996). However, as was argued in Chapter three, the confederal-intergovernmental structure of the EU makes its chain of policy-making and implementation considerably more complex than those of most political groupings (Blacksell, 1994; Archer and Butler, 1996). In essence, the process of EU environmental policy is comprised of five main stages; the determination of general aims and principles, the formulation of strategic Environmental Action Programmes (EAPs), the negotiation of specific legislation, their transposition into national law (when implemented as directives as opposed to regulations), and policy implementation within the Member States (Bailey, 1999a).

Thus far, the thesis has discussed the first three elements of this process in a relatively general manner. The purpose of this chapter is to explore the way in which policy formulation, transposition and implementation were managed in the specific case of the Packaging Waste Directive. The analysis begins by discussing the negotiation and transposition of the Directive, with particular reference to the EU's institutional dynamics and the resolution of conflicts between the Union's economic and environmental priorities. It then explores the strategies used by two Member States - Britain and Germany - to implement the Directive and assesses the extent to which their Packaging Waste Management Systems (PWMSs) have established environmentally- and economically-efficient methods of waste management. The chapter concludes by evaluating the development of recycling infrastructure and markets in the two countries.

5.2 The Negotiation and Transposition of the Packaging Directive

The Packaging and Packaging Waste Directive (94/62/EC) was formally agreed by the Council of Ministers in December 1994 and establishes, at its most basic level, targets

for the recovery and recycling of packaging waste within the Member States¹. By 2001, each must introduce systems which ensure that 50-65% of the packaging produced or imported into its territory is recovered, and that 25-45% is recycled or composted (Official Journal of the European Communities (OJEC), 1994). Articles 4 and 5, though abstaining from setting mandatory standards, specify that these systems should also promote the reduction and re-use of packaging waste², whilst Article 11 imposes further limits on the permissible concentrations of heavy metals in packaging. In common with most EU environmental legislation, the Packaging Directive contains several derogations, some general and others to meet the needs of specific Member States (Krämer, 1991). First, any Member State may introduce recovery and recycling targets above those contained in the Directive, provided these can be achieved without obstructing EU trade or hindering other states' compliance. At the other end of the spectrum, Greece, Ireland and Portugal, by virtue of their 'large number of small islands...rural and mountains areas and ... low level of packaging consumption' (OJEC, 1994: 14) are only required to recover 25% of their packaging waste by 2001 and may delay full compliance until 2005.

EU directives are so called because they direct Member States to legislate or take other effective action. This means they usually specify the standards Member States must achieve, not the methods used to reach the end result (Jordan, 1999). However, the Packaging Directive provides greater guidance than is customary for EU environmental legislation. It states that: 'Acting on the basis of the relevant provisions of the Treaty, the Council adopts economic instruments to promote the implementation of the objectives set by this Directive' (OJEC, 1994: 16). It also stipulates that all Member-State laws and implementing systems should observe the EU's environmental principles (Article 15) and must not obstruct the effective operation of the Single Market (Article 18). Within this framework, however, decisions as to the format and integration of economic instruments are left almost entirely to Member-State discretion. Therefore, although broad legislative harmonisation was one aim of the Directive, it is

¹ As noted in Chapter one, the term recovery denotes the collection of packaging waste for the purposes of recouping some form of value. Recycling is defined as, 'the reprocessing in a production process of the waste materials for the original purpose or other purposes including organic recycling but excluding energy recovery,' (OJEC 1994: Article 3, para. 7).

² The importance of waste minimisation to EU environmental policy was also established in the Commission's 1996 *Communication on the Review of the Community Strategy for Waste Management* (COM 96(399) final) (CEC, 1996d) and the Fifth EAP (Öko-Institut, 1999).

clear that equal value was placed on a flexible and locally-responsive implementation process.

The broad range of derogations and special provisions were, in fact, policy manifestations of the strong disagreements which occurred during the Directive's negotiation (Golub, 1996). These stemmed less from the draft Directive's advocacy of economic instruments and more from concerns about its economic impact and the lack of scientific basis behind the Commission's targets. The Commission's original proposals were resolutely opposed by a coalition of Council members (Britain, Spain, Ireland, Greece and Portugal), which objected to the introduction of mandatory reduction and re-use targets as well as the draft Directive's 'excessively' high recycling rates (see Table 5.1). These measures were generally supported by another grouping of environmental 'leader' states, headed by Germany (Golub, 1996). Because the coalition opposing the Environment Directorate's proposals was sufficiently large to prevent the formation of a qualified majority in the Council, many of the contentious items were either abandoned or diluted (Table 5.1) (European Parliament, 1994). In his analysis of the institutional dynamics of the negotiation process, Golub (1996) argues that the eventual format of the Directive demonstrates the power retained by Member States over policy formulation and, more specifically, by whichever grouping holds sway in the Council. He contends that this democratic power was not absolute, however, as the Commission and EP, along with the minority group within the Council, exerted sufficient influence that policy commitments were retained which some Member States would not voluntarily have countenanced. Although these concessions were ultimately necessary to maintain an accord in the EU process, one frustrated Parliamentary deputy branded the final Directive a 'mess of ill-assorted, inconsistent compromises' (European Parliament, 1994: 12). Nonetheless, a compromise was reached which permitted each Member State a degree of implementation flexibility within the broad framework of approximated EU standards.

Following agreement in Council, the next stage was to transpose the Directive into national law, a process the Commission has pursued vigorously. In 1998, six Member States received 'reasoned opinions' for failing to transpose the Directive fully. The Commission's first rebuked Britain for not transposing the Directive's 'Essential Requirements' (the general commitments to waste prevention and re-use); in response,

Table 5.1 Development of the Packaging Waste Directive at the EU

	<i>Comm 1</i>	<i>Comm 2</i>	<i>EPEC1</i>	<i>EP1</i>	<i>Comm 3</i>	<i>Common Position</i>	<i>EPEC2</i>	<i>EP2</i>	<i>Adopted</i>
Per capita limits	Yes	No	No	No	No	No	No	No	No
Minimum use of recycled materials	No	No	Yes	Yes	No	No	No	No	No
Hierarchy of preferred disposal	Yes	No	Yes	Yes	No	No	Yes	No	No
Opt-ups	No	No	Very broad	Limited	Limited	Very limited	Very limited	Very limited	Very limited
Derogations	No	No	Very limited	Very limited	Very limited	Yes	Yes	Yes	Yes
Five year targets									
• total recovery rate	60%	No	60%	60%	60%	50-65%	50%	50-65%	50-65%
• total recycling rate						25-45%	25%	25-45%	25-45%
• recycling rate per material	40%	No	40%	40%	40%	15%	25%	15%	15%
Ten year targets									
• total recovery rate	90%	90%	90%	90%	90%	No	No	No	No
• recycling rate per material	60%	60%	60%	60%	60%	No	No	No	No
• maximum landfill & incineration	10%	10%	10%	10%	10%	No	No	No	No
• heavy metals ban	No	No	Yes	No	No	No	No	No	No

Notes: *Comm 1* = Environment Directorate's pre-draft objectives
Comm 2 = Commission draft directive (12.10.92)
EPEC1 = First report by the Environmental Committee of the European Parliament (8.6.93)
EP1 = First reading by the European Parliament (23.6.93)
Comm 3 = Revised Commission proposal (9.9.93)
EPEC2 = Second report by the Environmental Committee of the European Parliament (7.4.94)
EP2 = Second reading by the European Parliament (4.5.94)

Source: Golub (1996: 323)

the British government introduced its own Packaging (Essential Requirements) Regulations (Department of the Environment, Transport and the Regions (DETR), 1998b). Belgium, Ireland and Portugal received reasoned opinions for similar transgressions (ENDS, 1997a), whilst Luxembourg and Greece were admonished for failing to adopt legislation by the required deadline (VALPAK, 1998: 12; Commission of the European Communities (CEC), 1998b). The Commission was equally intolerant of non-agreed exclusions from the Directive. Here, Britain and Finland were cited over their failure to transpose the Directive in Northern Ireland and Aaland respectively (CEC, 1998c; ENDS, 1997a). Whilst both regions have special status under the EU treaties (the reasons cited for the delays), Britain subsequently introduced legislation for Northern Ireland, though an application has been made to the ECJ to begin non-compliance proceedings against Finland (ENDS, 1997a).

There are two more fundamental cases outstanding at the time of writing. These concern the Commission's renewed attempts to overturn, first, a ban by Denmark on non-refillable beverage containers and, second, Germany's re-fill quotas (reviewed in Section 5.3.2). Aside from these cases, incomplete transposition of the Directive has generally been limited to those Member States that have experienced problems gaining domestic agreements on the design of implementing mechanisms and the apportionment of legal responsibilities. In the case of Luxembourg and Ireland, this has been further complicated by the fact both countries export most of their recyclable waste and, thus, need to develop secondary reprocessing agreements. Aside from the German and Danish cases, which raise fundamental issues about the relative priorities of environmental protection and EU free trade policies, this vigorous round of proceedings demonstrates the Commission's determination to ensure that national measures conform to the Directive's specified parameters.

5.3 National Packaging Waste Management Systems

5.3.1 Introduction

Although each Member State has developed its own distinct method for implementing the Directive, essentially four basic models of compliance exist; (i) the UK's Producer Responsibility Obligations (Packaging Waste) Regulations 1997 (the Regulations), (ii) Germany's *Verpackungsverordnung* (the Packaging Ordinance), (iii) the voluntary agreements established under the Dutch Packaging Covenant, and (iv) the Danish

system of integrated waste management. For reasons explained in Chapter four, the study concentrates on the British and German models and only considers other national systems where they are directly relevant to the study's main themes. A summary of national measures is provided in Table 5.2, but for a fuller account see Hagengut (1997) and Perchards (1998).

A variety of primary and data sources were used to analyse the German and British PWMSs. Secondary data consisted mainly of parliamentary debates, government consultations, submissions from industry to government reviews, consultants' reports and specialist media articles, whilst primary data were derived from interviews with representatives of the packaging industry and the survey of UK accredited reprocessors. The section begins by outlining the structure of the British and German PWMSs, then investigates the use of market-based and 'command-and-control' policies by national policy-makers. Finally, the performance of the two PWMSs are assessed in terms of the reprocessing rates and infrastructure development achieved in each country.

5.3.2 The German Model

The German Packaging Ordinance holds a fundamental place in the evolution of European packaging legislation, not least because Germany's domestic policies generated the push-pull dynamic which culminated in the creation of the Packaging Directive (London and Llamas, 1994). The 1991 Ordinance, described by Waite (1995: 137) as: 'the most prescriptive and demanding piece of environmental legislation passed by any European government with regard to packaging waste,' had serious implications for EU free trade because it introduced stringent packaging laws at a time when there was little EU legislation directly related to packaging waste. The Commission was therefore compelled either to challenge the Ordinance's legitimacy - though France, amongst others, was following the German lead (von Wilmowsky, 1993) - or to introduce harmonising European legislation. Considering the scale of Europe's waste management problem, its prominence in the EAPs, and the potential threat to free trade in the Single Market, several influential states saw EU legislation as the more progressive option.

Table 5.2 National Measures to Implement the Packaging Waste Directive

<i>Member State</i>	<i>2001 target (% of packaging weight)</i>			<i>Packaging Re-use Provisions</i>	<i>Packaging Refill Quotas</i>	<i>Tax on single trip containers</i>	<i>Landfill tax</i>	<i>Producer Responsibility</i>
	<i>Reduction</i>	<i>Recycling</i>	<i>Recovery</i>					
Austria		10-70 a	80	Some	Yes	No	n/a	Yes
Belgium		50	80	Yes	Yes	Some	n/a	Yes
Denmark		25-45	50-65	Yes	Yes	Yes	Yes	Yes
Finland	6	42	61	Yes	No	Yes	Yes	Yes
France		25-45	50-65	No	No	No	Yes	Yes
Germany		45	65	Yes	Yes	Yes	n/a	Yes
Greece		?	30	No	No	No	n/a	
Ireland		25	33	Some	No	No	Yes	Yes
Italy		25-45	50-65	No	No	No	Yes	Yes
Luxembourg		45	55	Yes	Yes	Yes	n/a	Yes
Netherlands	10	45 b	65	Yes	Yes	No	Proposed ban for packaging	Yes
Portugal		?	25	Yes	Yes	No	n/a	Yes
Spain	10	25-45	45-65	Some	No	Some	n/a	Yes
Sweden		30-90 a	70	Some	No	Repealed 1993	n/a	Yes
UK		16 c	52	Very limited	No		Yes	Yes

a Separate targets set for each packaging material covered by national legislation

b The second Dutch Packaging Covenant set an overall recycling target of 65% for 2001. This is a voluntary agreement, however, not part of binding legislation

c The UK recycling target is a minimum rate for each material covered by the Producer Responsibility Regulations.

Adapted from: Perchards (1998)

In addition to establishing ambitious recycling targets (Table 5.2), the Ordinance makes manufacturers and distributors responsible for the recovery and recycling of their packaging waste outside the public waste disposal system. In order to achieve this, the Ordinance stipulates that distributors must either remove secondary packaging from goods before offering them for sale or provide in-store facilities for consumers to leave their used packaging (Perchards, 1998; Raymond Publications, 1998). The federal government's desire to promote stringent environmental standards is also reflected in the Ordinance's interpretation of the waste management hierarchy. Whilst the reduction, re-use and recycling of packaging waste were made key objectives of the policy, incineration (the lowest element of the hierarchy before landfill disposal) was banned as a method of waste recovery. Finally, the Ordinance provides for the automatic introduction of a deposit-refund system for beverage containers if the national market share of re-fillable containers falls below 72% in any given year. This ruling was applied in each German state (*Land*) and permits individual *Länder* governments to impose mandatory deposit systems in their territories should re-use quotas not be met (Michaelis, 1995).

However, these obligations are waived for manufacturers and distributors taking part in an industry-organised system for collecting, sorting and recycling used packaging, the *Duales System Deutschland* (Dual System or DSD). This concession only applies to sales packaging, however; the recovery and recycling of secondary and transport packaging must be independently organised by obligated businesses (Michaelis, 1995). Although negotiations for the formation of the DSD were concluded before the adoption of the Ordinance, in reality, industry was offered little alternative except the reinstatement of the Ordinance's take-back and deposit-refund provisions (Haverland, 1999). The federal government therefore used its coercive powers to the maximum in order to 'persuade' industry to co-operate with its environmental agenda. 95 companies from the retail, consumer goods and packaging sectors originally formed the DSD. By 1998, the number of shareholders had increased to 600 (Whiston and Glauchant, 1996) and over 17,000 businesses now use DSD's systems to discharge their recycling obligations (DSD, 1998).

The DSD's main function is to organise a private network for the collection and sorting of sales packaging waste, based on collection plans agreed with each *Land* government, and using a mixture of kerbside collection (45% of DSD waste), bring schemes (27%),

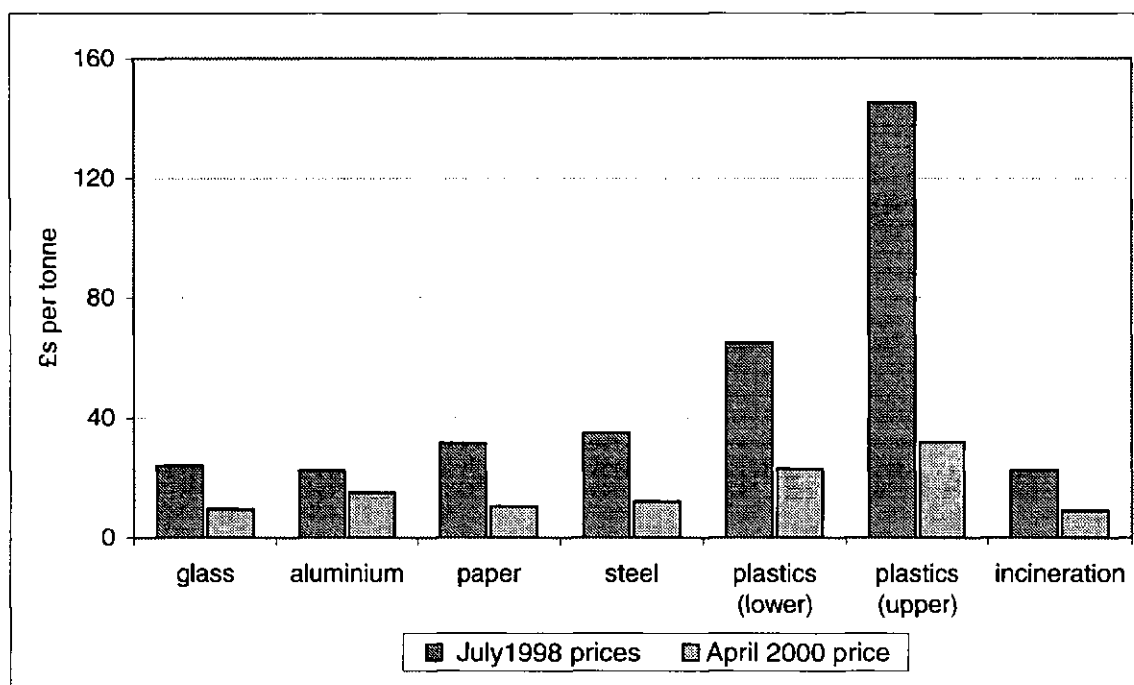
and combined schemes (28%)³. Its operations are financed by licence agreements with packaging suppliers to use the DSD's *Grüne Punkt* (Green Dot) logo on their packaging. This label is primarily designed to inform consumers that the packaging in question belongs to a business participating in the Dual System and that it should be separated ready for collection by the DSD's contractors. In order to obtain this licence, product manufacturers must pay a fee for each unit of packaging bearing the Green Dot, regardless of whether it is recycled. The manner in which fees are calculated is complicated but, broadly speaking, it takes account of the weight, area, volume and materials used in the packaging unit (see Figure 5.1). Funds from the Green Dot are then used to finance the DSD's collection systems as, with the exception of plastics, the licence fee only covers recovery costs. For more profitable materials, the resale value of reprocessed materials means that subsidies are not necessary, whilst for others reprocessing costs are allocated directly to the respective packaging producers (Michaelis, 1995). The final link in the recycling chain is provided by the DSD's guarantors. As part of the agreements to set up the Dual System, each materials sector established a unitary recycling association whose role is to guarantee the reprocessing of materials collected by the DSD in accordance with the terms of the Ordinance (Michaelis, 1995). A summary of the flow of packaging and funds associated with the Dual System is shown in Figure 5.2.

The original intention of the Dual System was therefore that the DSD should co-ordinate the actions of industries involved in the recovery and reprocessing of packaging waste. However, in its early years, this relationship was not particularly co-operative and the DSD came close to financial breakdown. Eichstädt *et al.* (1999) attribute this to four problems. First, some manufacturers using the Green Dot were free-riding the system by either under-declaring or failing to disclose the volume of packaging materials being put through the Dual System. Second, the licence fees in force at the time were too low to finance both the collection and reprocessing of waste packaging. As the accent of DSD was on collection networks, this led to a shortage of reprocessing capacity in Germany and the enforced export of large quantities of packaging waste to other EU states, often for landfilling (Waite, 1995; Michaelis, 1995). This not only destabilised materials prices in Europe but also undermined the

³ Personal correspondence with the DSD. Recycling studies generally agree that public participation is maximised under kerbside collection schemes (Pelletier *et al.*, 1996) except in rural areas, where increased travel discourages public involvement (Powell *et al.*, 1996).

political credibility of the German government and the entire concept of an industry-led Dual System (Lister, 1996, House of Commons, 1997). Third, the high targets imposed by the Ordinance forced the DSD to accept almost any contract it was offered and enabled some disposal firms to charge exorbitant collection fees. This crisis was further compounded when the guarantor for plastics recycling went into bankruptcy in 1993 and cited the lack of reprocessing finance from the DSD as the cause of its demise.

Figure 5.1 Green Dot Licence Fees

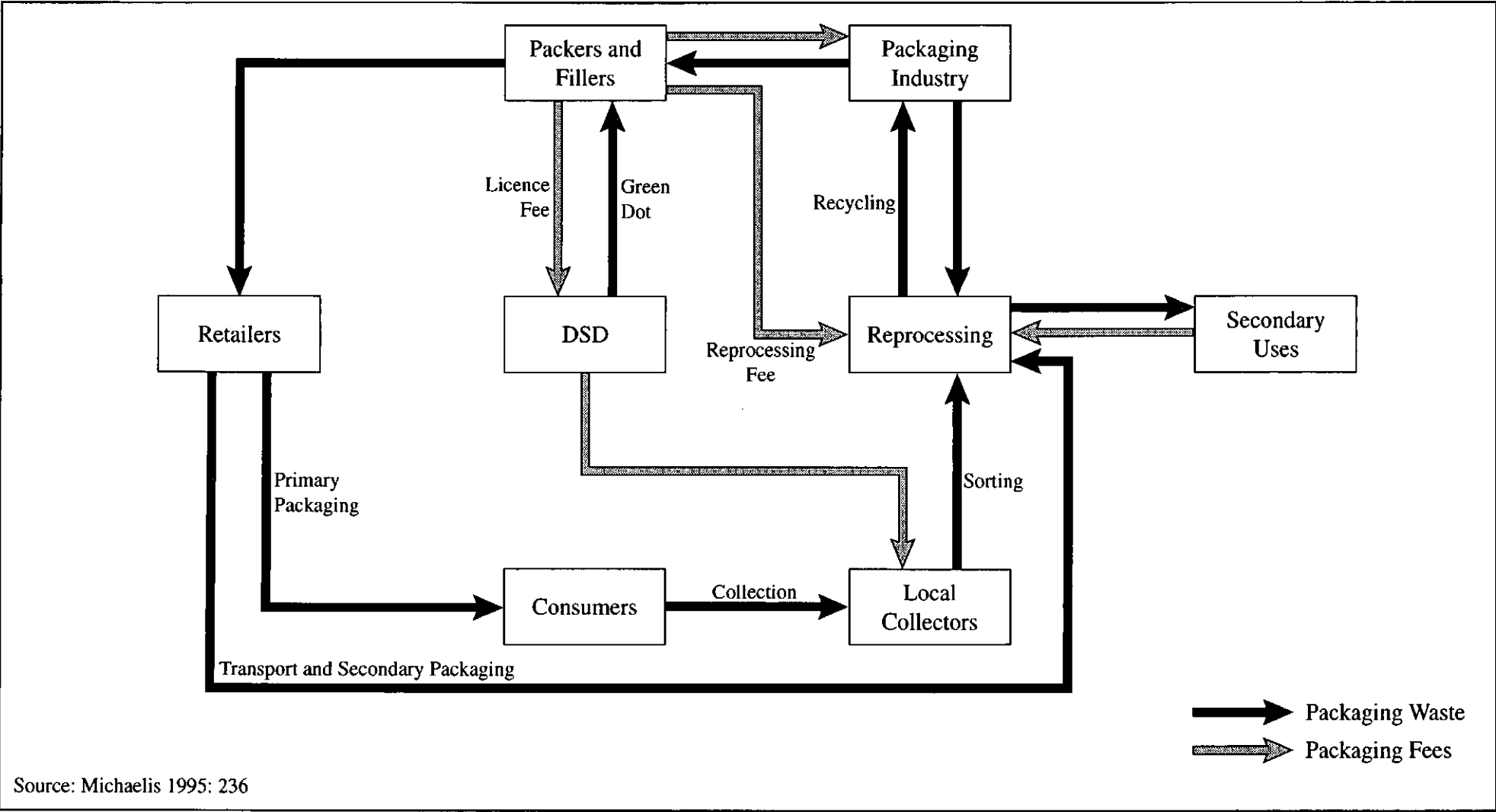


Item Fee in Pfennigs (including statutory VAT)

<i>Volume item fee</i>		<i>Area item fee</i>	
< 50-200 ml and > 3 g	0.1-0.6	< 150 - 300 cm ² and > 3g	0.1-0.4
> 200 ml - 3 litres	0.7-0.9	> 300 - 1600 cm ²	0.6
over 3 litres	1.2	over 1600 cm ²	0.9

Source: DSD (1998: 11)

Figure 5.2 Structure of the German Dual System



Source: Michaelis 1995: 236

To address this problem, a reconsolidation plan was developed with the assistance of the German Environment Minister (the *Töpfer* Plan). The main elements of this were the conversion of DSD's debts into long-term credit, the introduction of increased Green Dot licence fees, and the foundation of a new plastics guarantor (*Deutsche Gesellschaft für Kunststoff-Recycling mbH (DKR)*) by the DSD in conjunction with two German energy suppliers (Eichstädt *et al.*, 1999). As a result of the programme, the DSD has reduced waste exports to 19% of all packaging waste produced in Germany (DSD, 1998) and has invested heavily in new recycling technology and the reduction of expensive exports. These actions stabilised the DSD's financial position to the extent that it recently announced a 9.5% reduction in the Green Dot fee, effective from January 1999⁴. The success of the reconsolidation plan can largely be attributed to three factors; the government's desire not to repeal the Dual System concept, the influence of powerful retailers seeking to avoid the re-imposition of the Ordinance's more constrictive provisions and, finally, concessions from disposal firms who feared losing lucrative recycling contracts (Eichstädt *et al.*, 1999). Therefore, although the Dual System has subsequently succeeded in fostering co-operation between industry sectors, the main impetus came from the government's ability and willingness to resort to command-and-control regulation.

Although Germany's Packaging Ordinance began the process which culminated in the EU Directive, certain aspects of its legislation have been repeatedly challenged by the European Commission. In 1995, before the transposition deadline for the Directive, the Commission argued that Germany's re-fill quotas added disproportionate transport costs to drinks importers and therefore transgressed Article 30 of the Treaty (Perchards, 1998). In its defence, Germany cited Life Cycle Analysis (LCA) studies produced by the Fraunhofer Institute, which supported re-use systems over recycling, and argued that the restriction on free trade was justified under the environmental protection requirements of the Treaty (Otto, 1999). The Commission decided not to pursue the complaint at the time, mainly because there was no direct EU legislation on the matter⁵. Since the Packaging Directive came into force, however, there have been renewed efforts to overturn the German quota scheme. Several Member States, including Britain and Sweden, lodged formal objections in 1998, claiming that though the

⁴ Personal correspondence with DSD.

⁵ This precedent had been established in the Danish Bottles Case in 1988 (302/86, ECR 4607). Here the ECJ affirmed the environment as an essential part of the Treaty and the right of Member States to enact legislation in the absence of satisfactory EU measures.

Directive accords equal priority to prevention, re-use and recycling, the Ordinance discriminates against recycling by granting preference to re-use systems (ENDS, 1997b; 1997c). At the same time, the German government came under domestic pressure to oppose any weakening of re-fill quotas during negotiations to amend the Ordinance in 1998. Following the rejection of several draft versions by the *Länder*-based *Bundesrat* (ENDS, 1998b; 1998c), mandatory deposits were retained but will now only be invoked should quotas not be achieved for *two* consecutive years. In fact, the German government recently informed industry representatives that because re-use quotas were not met in 1998 and 1999, mandatory deposits will be triggered from June 2001 (ENDS, 2000a). The Commission has responded to the realisation of what was previously only a latent threat by instituting proceedings against Germany (ENDS, 2000b)⁶. The other major amendment in the 1998 Ordinance was its relaxation of the ban on Energy from Waste (EfW) incineration for transit and sales packaging made from directly 'renewable' materials (Perchards, 1998).

Therefore, whilst Germany was instrumental in defining the agenda of EU packaging waste legislation, it has come under pressure to harmonise its systems with those of other Member States. As with many other areas of policy, the disagreement has centred on the extent to which environmental protection should take precedence over EU free trade. Though such disputes are almost inevitable considering the potential areas of conflict between the two policy objectives, they highlight some of the practical difficulties involved in integrating environmental protection into the EU's wider agenda (Bailey, 1999a). Nevertheless, the German model remains the most mature in the EU and has provided important lessons for other Member States. Not only have other states been more circumspect about setting ambitious recycling targets and implementation time-frames⁷, they have also concerned themselves more keenly with establishing viable reprocessing infrastructures. Probably the most important insights from the German Ordinance, however, have been into the trade-offs which exist between environmental and economic efficiency (Brisson, 1993). Whilst there are signs that Germany is moving slightly away from command-and-control packaging

⁶ The ENDS report notes that this was after a long period of soul searching by the Commission, which had been under intense pressure from the packaging industry. Intriguingly, the move was opposed by the Commission's Environment Directorate, whose waste unit is headed by Ludwig Krämer, a German environmental lawyer known for his 'green attitudes' and refusal to compromise on matters of legal principle (Jordan, 1999:77)

⁷ The schedule to implement the Packaging Ordinance was just 18 months (though Austria opted for a 12 month implementation programme). By contrast, the UK legislation is being phased in over four years.

waste policies, the political capital invested in DSD by the German government and its desire for high environmental standards has necessitated a continuing emphasis on strong and sometimes constrictive regulatory leadership.

5.3.3 The British Model

Negotiating the UK Regulations

Although the British government did not introduce packaging waste legislation until after the Directive was adopted, preparations began soon after the German Ordinance came into effect⁸. The main thrust of government policy at that time was to explore implementing mechanisms that could achieve the objectives of the Directive 'in a manner which is efficient, equitable and least burdensome' (Department of the Environment (DoE), 1996b: Ministerial foreword). The foundation for the DoE's approach was the development of a comprehensive dialogue with industry on the format and implementation of a business-led scheme for the recovery and recycling of packaging waste. However, this co-optive challenge was accompanied by the threat that a unilateral solution would be imposed by the government if industry failed to produce a suitable plan (Haverland, 1999).

A number of working parties were established to assist these discussions. Following initial forays by INCPEN and COPAC⁹, the government commissioned the Producer Responsibility Group (PRG), a forum consisting of 26 businesses concerned with packaging issues, to prepare a framework plan for packaging recycling and recovery. The PRG's draft proposals, submitted in February 1994 and establishing principles which would form the future PWMS, supported a scheme encompassing all sectors of the packaging chain, the creation of a competitive, cost-effective recycling market, and the use of price-based measures as an incentive for packaging optimisation (Figure 5.3). On completing its report, the PRG was de-commissioned and replaced by another packaging chain organisation, VALPAK, and its Working Representative Advisory Group (V-WRAG) (ENDS, 1995a).

⁸ For example, the DTI and DoE commissioned two studies by Environmental Resources Limited to explore methods for recovering resources from waste (DTI/DETR, 1991; 1992).

⁹ Industry Council for Packaging and the Environment, and the Consortium of the Packaging Chain, respectively.

Figure 5.3 PRG Draft Proposals, February 1994

- i. On the basis of current data, recovery of around 58% of the UK's packaging waste (as against the 30% currently being achieved) is achievable by the year 2000, but not on a voluntary basis. Underpinning legislation was required to assure compliance and to provide the necessary incentive to create business operated schemes to organise recovery and recycling;
- ii. All parts of the packaging chain need to be involved, from raw materials manufacturers to retailers, if effective co-operation is to be achieved and recycling costs minimised. It is essential that business sectors co-operate to increase end-use markets for recyclate and to cause investment in new reprocessing capacity while retaining a market led approach^a;
- iii. There is a need for renewed commitment to waste to energy which is more appropriate than recycling for some packaging waste;
- iv. There is a need for incentives for minimisation, for example through material-specific charging;
- v. There is a need for continuing consumer awareness and participation.

a The sectors of the packaging chain, as defined by the PRG and the DoE, are (i) raw materials producers, (ii) converters (manufacturers of packaging), (iii) packer-fillers (manufacturers of products which use packaging), (iv) wholesalers, and (v) retailers. The collective term for these companies is packaging producers.

Adapted from: DoE (1996b: pages not numbered)

Despite the PRG's recommendation of a shared approach, the government initially preferred the administrative simplicity of focussing all responsibilities on one sector of the packaging chain (ENDS, 1995b). Even though all companies participating in V-WRAG favoured some form of shared responsibility, they initially failed to agree a common proposal. Retailers generally supported a single onus on converters and the use of market forces to diffuse costs and responsibilities to other sectors of the chain (ENDS, 1995b), whilst others favoured a variety of multi-point options. The choices considered by V-WRAG and its final proposal are summarised in Figure 5.4.

Figure 5.4 V-WRAG Proposals for Packaging Recovery, 1995

V-WRAG Draft Proposal

- i. **Single Point Obligation:** One point of legal obligation for packaging recovery and recycling (either converters, packer-fillers or wholesalers/retailers), with market forces ensuring all sectors of the chain contribute. This option was favoured for its simplicity and low implementation costs, but raised concerns over the ability of sectors further up the chain to pass on compliance costs to consumers.
- ii. **Omni-Point:** Targeting of 'brand owners' for each product as a means of placing product stewardship obligations with those primarily responsible for generating the packaging. This scheme was seen as complicated and costly to administer.
- iii. **Combined Industry Scheme:** Obligation on the 'first purchasers of packaging for use,' primarily packer-fillers.
- iv. **Multi-Point:** Obligation on all sectors of the packaging chain and the allocation of an appropriate share of the recovery targets in relation to the packaging each handles. It emphasised the idea of businesses joining an industry-wide compliance scheme to manage recovery and recycling activities.
- v. **Equi-point:** A basic requirement on all sectors of the packaging chain to recover packaging waste arising on their premises, with additional obligations upon specific sectors (packer-fillers and retailers to collect household waste, converters in terms of recycled content in packaging, and raw materials manufacturers and reprocessors to reprocess or valorise collected waste).

Source: ENDS (1995b: 37-8)

The Shared-Responsibility Concept

- i. Legal duty on all companies to ensure that packaging waste arising on their premises is valorised to agreed levels.
- ii. For packaging supplied further down the packaging chain or to end users, a duty of care to take all reasonable measures to ensure the valorisation of this waste packaging.
- iii. The establishment of a collective scheme (VALPAK) to manage recovery and valorisation responsibilities on behalf of obligated companies.
- iv. Companies choosing to manage their legal duties individually rather than by joining a collective scheme should be required to submit an annual report to the Environment Agency to demonstrate their PWM plans. Failure to do so should be made an offence and the Environment Agency should have the power to require businesses not presenting convincing plans to make good any deficiencies.

Source: ENDS (1995b: 38)

Whilst V-WRAG's preference for 'shared responsibility' and a non-quantified duty of care across the packaging chain gained widespread industry support (ENDS, 1995c), the DoE insisted on clear and binding recovery targets (ENDS, 1995d). In the end, it was the threat of an imposed solution that forced industry's acceptance of mandatory recovery and recycling obligations at a meeting convened by the DoE on 15 December

1995. The targets and responsibilities agreed at this meeting are shown in Table 5.3. The DoE, disgruntled with the belligerent attitude of V-WRAG, decided that wider industry consultations and the drafting of legislation should be overseen by a more neutral body and appointed a trusted emissary, Sir Peter Parker, to head the newly formed Advisory Committee on Packaging (ACP). Following its initial work, the UK Regulations were finally brought before the House of Commons in 1996 (DoE, 1996b).

Table 5.3 Targets and Responsibilities in the 1997 UK Regulations

	<i>Recycling (%)</i>	<i>Recovery (%)</i>
1998-1999	7	38
2000	11	43
2001	16	52
<i>Sector Responsibilities</i>		<i>% of Recovery and Recycling Targets</i>
Raw materials manufacturer		6
Converter		11
Packer-filler		36
Retailer		47
Wholesaler		83

All figures are expressed as percentages of the total packaging produced or imported into Britain but exclude exports. Packaging materials covered by the UK Regulations are paper/board, glass, steel, aluminium, plastics and, from 2000, wood.

Source: DoE (1996b: pages not numbered)

Though accounts of these negotiations do vary - one commentator, for example, claimed the UK government was 'pathetically anxious not to upset industry and provided no leadership' - the general view is that industry played an important but defensive role during the policy's formulation. Whilst the *desirability* of a business-led scheme was supported from the outset, industry's desire for a purely voluntary approach provided the government with insufficient guarantees of compliance with the Directive. Realising that failure to agree a common approach would risk the abandonment of the partnership approach altogether, industry was forced to concede ground. The negotiation of the UK Regulations therefore corroborates Lévêque (1995)

and Whiston and Glachant's (1996) view that voluntary industry environmental agreements are partially illusory because they are usually formed to fend off coercive pressure from government. The more powerful sectors in industry were nonetheless able to reduce their absolute losses by conceding the legitimacy of the initiative then engaging defensively in the formulation process (see Chapter seven).

Producer Responsibility and the PRN System

In accordance with the agreements reached in 1995, the Producer Responsibility Obligations (Packaging Waste) Regulations 1997 impose legal responsibility for the recovery and recycling of packaging materials upon companies which manufacture, supply or sell packaging or packaged products (*Packaging Producers*) (DoE, 1997). Thus, companies which only use rather than supply packaging, are exempted from direct responsibilities under the Regulations. The 1997 Regulations were followed by the Packaging (Essential Requirements) Regulations 1998 (DETR, 1998b). These require, first, that packaging should be manufactured in a manner which minimises its volume and weight and permits re-use or recovery and, second, that packaging should meet EU specifications on concentrations of heavy metals. As with the Directive, the 'Essential Requirements' contain no mandatory targets on waste minimisation or re-use.

From 1998, all companies with an annual turnover exceeding £5 million and handling over 50 tonnes of packaging are required to submit annual returns to the Environment Agency proving their compliance with the sector targets contained in Table 5.3. Using these criteria, the number of obligated UK businesses was initially estimated at 5,000 (Perchards 1998), though the actual number registering with the Environment Agency in 1997 was 3,837 (Environment Agency, 1998a). This figure will increase to over 11,000 in the year 2000 when turnover thresholds reduce from £5 million to £2 million (DETR, 1998a).

However, two practical issues still needed to be addressed before the policy could be implemented. First, substantial investment in the reprocessing industry was required in order for Britain to meet EU requirements. Second, it was in everybody's interests that the PWMS minimised the financial and operational impact of the Regulations on industry, as the majority of companies classified as packaging producers did not wish to

be heavily involved in the expensive process of packaging recovery and recycling (DoE, 1996b; 1996c). To achieve these aims, two groups of organisations, *Compliance Schemes* and *Accredited Reprocessors*, became licensed to operate what is known as the *Packaging waste Recovery Note* (PRN) system.

Companies charged with recovering and recycling their packaging have two compliance options. First, they can apply for independent registration with the Environment Agency, SEPA¹⁰ or the Northern Ireland Heritage Service. Producers choosing this option are required to submit annual waste management plans to the agencies and provide evidence that they have discharged their obligations. In 1997, 33.6% of packaging producers chose this alternative (Environment Agency 1998a). Second, they can join one of several compliance schemes, organisations registered with the Environment Agency to manage producer-recycling networks. In return for membership fees and recovery and recycling charges to a scheme (sometimes termed a *materials levy*), producers can secure immunity from prosecution and, as with the Green Dot scheme, any practical involvement in the recycling process. Thirteen schemes are currently registered with the Agency, of which VALPAK, with over 2,000 members, is by far the largest (Environment Agency 1998a). Compliance schemes may act on behalf of as many producers as they wish, provided they can demonstrate to the Agency that their packaging recovery services are effectively organised. However, only a few schemes are themselves physical reprocessors of packaging waste. Instead most use materials levies to purchase recycling services from accredited reprocessors. As such, they are chiefly a means of aggregating producer-recycling obligations into more effective bargaining collectives. For this reason, the majority of the UK's packaging producers see the compliance-scheme method as the most cost-effective means of complying with the Regulations.

The principal role of the accredited reprocessors is to provide physical recycling facilities and services. In most cases, they are existing recycling companies that have registered with the Environment Agencies under the PRN scheme. The quantity of packaging waste reprocessed in the UK is monitored by the use of PRNs. PRNs are a tradable certificate issued by accredited reprocessors in respect of packaging materials delivered for recycling or energy recovery. The notes specify the weight and type of

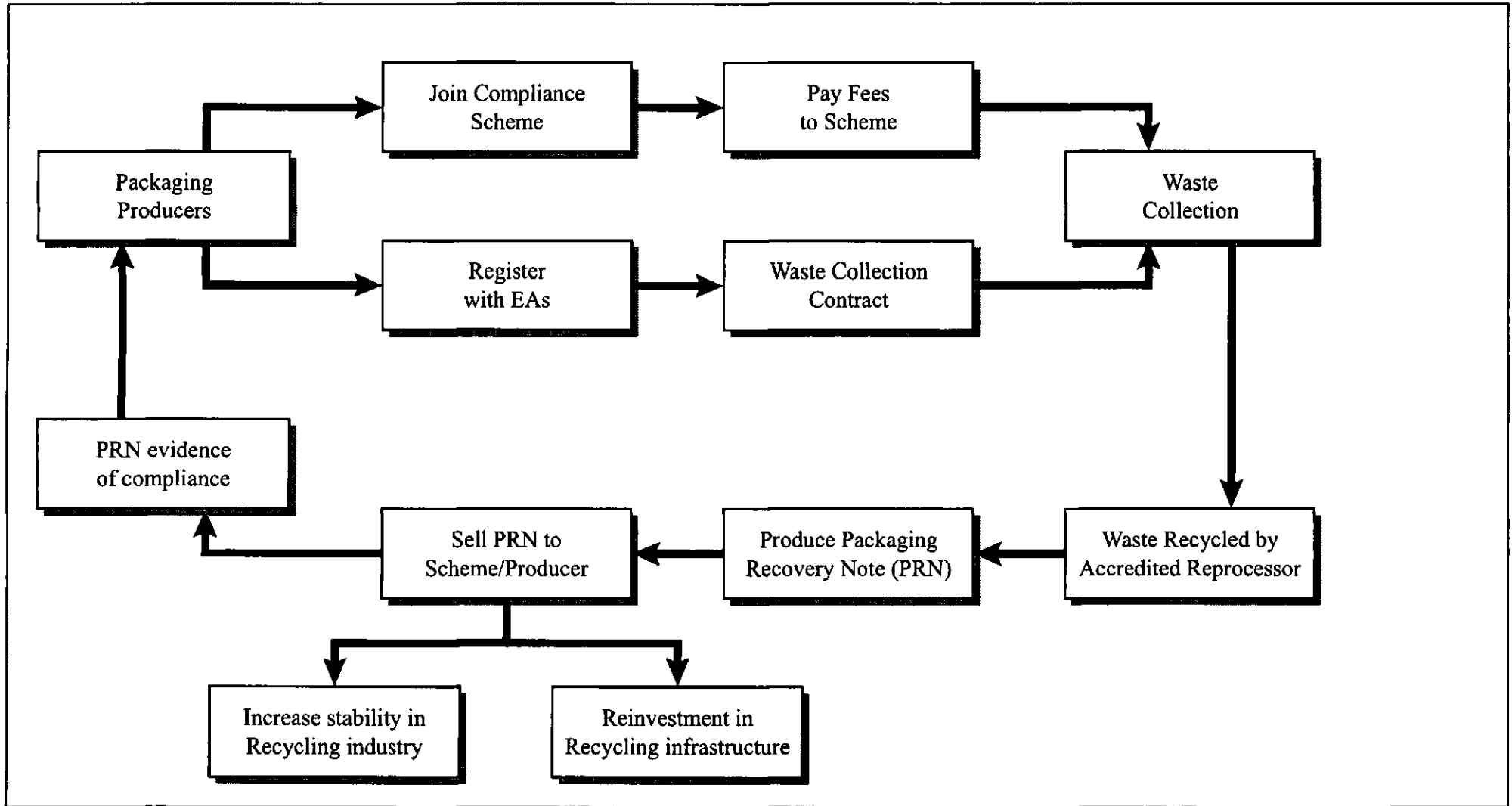
¹⁰ Scottish Environment Protection Agency.

packaging material processed as well as the reprocessing method used (ENDS, 1998d). Although the DETR provides detailed definitions of items that constitute packaging (DETR, 1997a, 1997b), PRNs may be produced from either industrial-commercial or household waste. Accredited reprocessors are then entitled to issue or sell completed PRNs to compliance schemes or producers; they in turn submit them to the agencies as evidence of compliance with the Regulations. A synopsis of the packaging waste and funding flows in the PRN system is shown in Figure 5.5.

Under the PRN system, therefore, packaging producers are required to pay for less environmentally damaging methods of waste management. However, it also provides means by which they can discharge their recycling duties - either through compliance-scheme membership or PRN purchases - without direct involvement in recycling. Furthermore, because the UK Regulations do not distinguish between industrial-commercial and domestic waste, most schemes have concentrated on collecting high volume, homogenous - and thus lower cost - industrial-commercial waste (Bailey, 1999b). This arrangement therefore fulfils one of industry's main requests, that the Regulations should implement the Directive in a cost-effective manner.

The PRN system also serves two purposes for accredited reprocessors. First, the additional revenue provides financial support for an industry whose development has historically been hindered by the volatility of basic materials prices (Bailey, 1999b; 2000). Second, the expectation is that surplus funds will be invested in new reprocessing capacity, an outcome the government has sought to encourage by allowing PRN prices to be determined by market forces rather than state involvement. According to market theory, prices should respond to the investment requirements of each reprocessing sector, which are in turn determined by the Regulations. However, the balance of market power is theoretically safeguarded by the aggregate bargaining power of compliance schemes and by competition in both the compliance scheme and reprocessing sectors (Bailey, 1999a, 1999b). Monopolistic behaviour is therefore prevented by market contestability, whilst competition has produced a system which is supposedly cost-effective and responsive to changing regulatory conditions. However, though the scheme contains 'financial carrots' for industry, it is underpinned by the understanding that constrictive legislation will be introduced if it fails to achieve EU targets (DETR, 1998a, DETR, 1999a).

Figure 5.5 Funding and Packaging flows in the UK PRN System



5.3.4 Comparing the UK and German Models

Perhaps the most striking feature of both the British and German arrangements for packaging recovery and recycling is the sheer complexity of the two systems. Whilst this undoubtedly reflects the nature of the task in question and makes comparisons difficult, the two schemes nonetheless have three common features. First, both Britain and Germany have framed their policies around the concept of producer responsibility, the idea that industry should be made accountable for the environmental stewardship of its products throughout their entire life cycle (Fenton and Sinclair, 1996). Within this framework, however, both governments have granted industry substantial leeway in the management of these responsibilities, the UK through its semi-autonomous packaging market and Germany under the Dual System. Finally, the PPP has been applied in both countries in the form of packaging waste charges for producers, but its diffusion to other polluting parties - generally speaking, the service sector and the general public - has been left to market forces (Bickerstaffe and Barrett, 1993). This theoretically enables both systems to charge consumers for the costs of packaging waste management without the government having to resort to unpopular public taxes (Bailey, 1997). Therefore, although previous commentaries have sometimes presented the British and German approaches to environmental policy as highly distinct (Ramus, 1991; Haigh, 1996), there are strong similarities in the way they have applied price-based regulation in this instance (Bailey, 1999a).

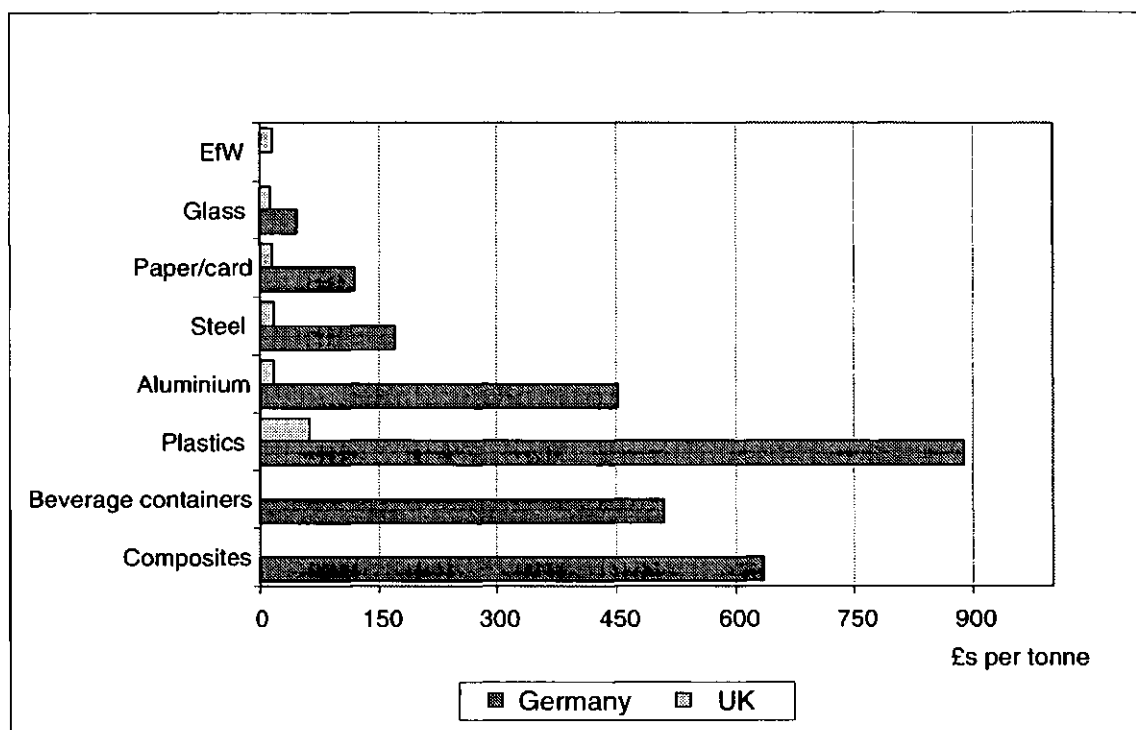
However, these similarities mask fundamental differences in the way each country has transposed and implemented the EU Directive. The key point of divergence in both areas has been the relative priority accorded to environmental protection and cost-effective compliance. During policy transposition, the UK government recognised that even the Directive's lower targets represented a major challenge to Britain's nascent recycling industry and took maximum advantage of the leeway granted in the Directive. By contrast, the derogations negotiated at the EU have enabled Germany to maintain its stringent recycling policies but, equally, it has endured a lengthy stand-off with the Commission over its re-use provisions for beverage containers. Similarly, even following the revised Ordinance's recent relaxation on EfW incineration, German standards are still largely informed by its zealous environmental stance. Conversely, the Regulations have made maximum use of this comparatively inexpensive means of recovering waste.

Other differences between the two countries initially appear more significant than they actually are. A key example is the apportionment of recycling responsibilities, where German policy has focused mainly on the actions of manufacturers and distributors but Britain has adopted a broader shared-responsibility concept. Though this could again be interpreted as an attempt by the British government to minimise the impact of the Regulations, in practice, German manufacturers and distributors have also passed a substantial proportion of their waste collection costs to suppliers. This has occurred to such an extent that many German raw materials and packaging manufacturers have resorted to paying waste management companies directly in an attempt to regain control of their costs¹¹. Similarly, many UK retailers have attempted to reduce their compliance costs by demanding that suppliers re-design their products and reduce their packaging requirements. This manoeuvre has caused considerable distortion of the sector responsibilities agreed in 1995 (ENDS, 1998d) and has led to a vociferous campaign by the converter sector for a 2% reduction in their recycling obligations (MRW, 1998a).

The balance between economic and environmental priorities in the two countries is also expressed in their respective use of economic instruments. The simplest but most revealing distinction is the point at which packaging waste charges are levied. Green Dot charges are calculated on the basis of the total packaging produced by DSD members regardless of whether it is recycled. In Britain, PRN charges are raised following the reprocessing of waste and therefore only relate to the percentage targets contained in the Regulation. This not only alters the relative costs of the two systems and the incentives created, it also influences the amount of revenue available for investment in infrastructure, innovation and public education. This difference is further reinforced by the rates at which recycling charges are set in each country (Figure 5.6). Based on current Green Dot fees (excluding area and volume fees) and average PRN prices between 1998 and 2000, German prices range from between 3.4 times those in Britain (glass) to 14.6 times (plastics) and 25.7 times (aluminium). Whilst this distinction is partly offset by the exclusion of non-sales packaging from the Dual System (though secondary and transport packaging are covered by the Ordinance), it suggests that the German system is primarily geared towards ambitious environmental targets and the British arrangements to cost-effective compliance with the Directive.

¹¹ Personal communication with Perchards.

Figure 5.6 Recovery and Recycling Charges, Germany and UK



The final distinction concerns the extent to which market-forces drive the British and German recycling schemes. As discussed earlier, the PRN system was specifically designed to reap the efficiency benefits accruing from competition and market-based pricing (Beesley and Littlechild, 1983; Gardner, 1996). At the time of writing, there were thirteen compliance schemes and 210 accredited reprocessors registered in Britain. Although there are a similar number of reprocessing companies in Germany (see Appendices 7a and 7b for details of their distribution), the DSD and its associated guarantors are responsible for co-ordinating their activities (Michaelis, 1995; DSD, 1998; Environment Agency, 1998b; 1998c; DETR, 1999a; MRW, 1999c). Thus, co-operation between industries has generally taken priority over the promotion of competition-led efficiencies. The different priorities in the two countries are further underlined by the methods used to calculate collection and reprocessing costs. Whilst PRN prices are fixed by market forces in response to legislative and competitive pressures, Green Dot charges are centrally determined by the DSD on the basis of, first, Life Cycle Analyses (LCAs) of the environmental impact of packaging waste and, second, the DSD's operational costs. Although it is dangerous to make more than provisional judgements about the environmental merits of each arrangement, the link to environmental effectiveness seems clearer in the German model, whilst that to

economic effectiveness is more apparent under the British system. That said, there is pressure for the DSD to adopt a more competitive regime. The amended Packaging Ordinance stipulates, first, that all packaging collection must be reprocessed under competitive conditions and, second, that waste disposal companies must publish their recycling costs in order that their competitiveness can be assessed (Flanderka, 1998).

5.4 The Development of Packaging Waste Management in Britain and Germany

5.4.1 Introduction

Having identified the important features of the British and German systems of packaging waste management, the next stage is to analyse their performance against the Directive's main objectives. In some respects it is difficult to compare the two schemes, as the Ordinance has been in force since 1991, whilst the Regulations are still very much in a transitional stage. The Dual System is therefore firmly embedded whilst the PRN system has yet to be fully tested. However, the British government and industry's involvement in monitoring the development of the PRN system has, as much as any predictions can, helped to make detailed comparisons possible. However, it is first necessary to identify the criteria used to evaluate each model. Three specific themes are considered in this chapter; the development of infrastructure for the reprocessing of packaging waste, the establishment of collection systems, and the efficacy of environmental charging systems in financing these developments (Sinclair and Fenton, 1997). The growth of end-use markets for recyclate and the success of environmental charges in changing polluter behaviour, the other main objectives of the Directive, are discussed in Chapters six and seven.

A combination of primary and secondary data sources were used for this analysis. The secondary data were derived from government documents, specialist press releases, consultancy reports and academic commentaries. The primary data were obtained from the survey of UK accredited reprocessors. A corresponding survey of the German reprocessing sector was not necessary because the information required was already available from secondary data sources. At the time the research was conducted, 133 separate reprocessing companies were registered with the Environment Agency, all of which were contacted for the survey. The response rate of 48 (36.1%), though slightly disappointing, can largely be attributed to the unwillingness of companies to disclose commercially sensitive information. However, the DETR conducted a similar review

in 1999 which provided more comprehensive data on the development of the UK reprocessing sector (DETR, 1999a). It was therefore decided that the DETR's quantitative data would be more representative but that two pieces of qualitative data from the reprocessor survey should be used in the analysis, details of partnerships developed by reproducers to encourage the collection of packaging waste and reprocessor opinions on the merits and problems of the PRN system.

5.4.2 Present and Predicted Reprocessing Capacity in Britain and Germany

In order to quantify the reprocessing capacity required in each country, the first task was to identify the volumes of packaging predicted to enter the British and German waste streams in 2001, the Directive's compliance deadline. The DETR's estimates for the UK, shown in Table 5.4, indicate that, with the exception of wood, steel and glass, consumption of packaging materials will increase by between 1% and 4% per annum in the period 1998-2001 (DETR, 1998a; 1999a). Whilst forward data was not available for Germany, packaging consumption excluding wood stood at 11.84 million tonnes in 1997 and has decreased by 16.5% since the Ordinance's introduction in 1991 (DSD, 1998). The DETR has also produced projections of expected growth in each reprocessing sector, based on data provided by UK materials organisations (Table 5.5). These figures suggest that the Regulations and the PRN system will encourage expansion in all sectors, but that there will be wide variations in this growth. However, the DETR also records that there was sufficient reprocessing capacity in all sectors to meet the recovery and recycling targets set for 1998 (DETR, 1999a).

Again, predictions of reprocessing capacity were not available for Germany though, as with the packaging consumption figures, data have been published for the period 1992-1997. Table 5.6 shows the recycling rates achieved for each packaging material and the performances achieved for sales and non-sales packaging though, for the purposes of this analysis, the total recycling rates for Germany are more comparable with those achieved in the UK. Whilst reprocessing rates for non-sales glass and plastics were significantly lower than for sales packaging, the general conclusion is still that the Dual System has exceeded the Directive's targets by some margin. However, two areas of concern remain, the high cost and environmental impact of packaging waste still being exported for reprocessing (Michaelis, 1995; Eichstädt *et al.*, 1999), and the amount of plastics waste being reprocessed using a process called 'feedstock recycling.'

Table 5.4 Packaging flowing into the UK Waste Stream, 1997-2000 (thousands of tonnes pa)

	<i>Actual</i>	<i>DETR growth estimates</i>	
	<i>1997</i>	<i>1998</i>	<i>2001</i>
Paper	3,611	4,000	4,308
Glass	2,185	2,200	2,200
Aluminium	155	109	111
Steel	833	735	735
Plastics	1,612	1,700	1,912
Total, excluding wood and 'other'	8,396	8,744 a	9,266
Wood	2,372	1,300	1,300
Other	968	200	200
TOTAL	11,736	10,244	10,766

a There has been considerable uncertainty over these figures, however. In 1998, obligated businesses reported only 6,965,962 tonnes (excluding wood and 'other'). This has obviously led to uncertainties as to the precise volumes of recovery and recycling required to meet the EU target.

Sources: DETR (1998a: 21), DETR (1999a: 10)

Feedstock recycling is a process whereby plastics are separated into their constituent elements and incinerated in blast furnaces to generate energy. The point of contention is that blast furnace reprocessing produces more harmful air emissions than normal EfW processes (which are restricted in Germany because of their environmental impact) (Eichstädt *et al.*, 1999). In an effort to curb the use of feedstock recycling, the 1998 Ordinance requires that at least 36% of plastic packaging should be mechanically recycled (Perchards, 1998). The DSD and the plastics guarantor, DKR, whilst defending the benefits of feedstock recycling, have also embarked on programmes to improve the viability of mechanical recycling and to reduce the environmental impact of feedstock processes (DSD, 1998).

The first evidence that the PRN system may fail to meet the targets set by the Regulations comes from data produced by the DETR's 1999 review (DETR, 1999a).

Table 5.5 Predictions of UK Recovery and Recycling Capacity 1998-2001

	<i>Expected performance (thousands tonnes)</i>		<i>% increase</i>
	<i>1998 actual</i>	<i>2001 estimate</i>	
Paper	1,888	1,921	1.7
Glass	658	730	11.0
Aluminium a	15	53	262.8
Steel a	183	235	28.8
Plastics	126	212	68.2
Wood	n/a	350	n/a
Total recycling	2,870	3,501	21.8
EfW and composting	448	726	61.9
Total recovery expected	3,318	4,227	27.4

a However, the aluminium and steel industries can reprocess up to 375,000 and 6 million tonnes annually respectively, should the materials become available. Currently 75,000 tonnes is reserved for aluminium packaging and 144,000 tonnes for steel.

Source: DETR (1999a: 21)

The DETR's projections (Table 5.7) are based on, first, the need to recycle 25% of the overall tonnage of packaging consumed in Britain and, second, the requirement that a minimum of 15% recycling should be achieved for each material. If these calculations prove accurate, the paper, glass, aluminium, steel and wood sectors will all meet the 25% target, but there will be an annual deficit of 266,000 tonnes for plastics by 2001 and one of 726,000 tonnes for EfW incineration. Moreover, as the DETR's figures are contingent on paper, glass, aluminium, steel and wood exceeding their 25% recycling quota and cross-subsidising other sectors, anything below this may increase the incineration deficit further. Theoretically, this could be as high as 1.9 million tonnes.

On the basis of these findings, the DETR report highlights factors which may inhibit the expansion of reprocessing in each material sector. Whilst a deficit is not predicted for either glass or paper, the DETR notes that both are international commodities and vulnerable to fluctuations in global demand and prices (also Hanley and Slark, 1994).

Table 5.6 Packaging Recycling Rates in Germany 1992-1997

	<i>Thousands tonnes recycled</i>					<i>Recycling rates 1997 (%)^b</i>		
	1992	1993	1995	1997	1998	Total	Sales	Non-sales
Paper	306	966	1,255	1,372	1,415	59	89	44
Glass	542	2,388	2,572	2,736	2,705	75	87	10
Aluminium	0	9	31	40	43	72	83	45
Steel a	29	250	259	312	345	82	73	98
Plastics	41	281	504	567	600	45	69	14
Composites b	5	52	297	420	345			
Total	923	3,946	4,918	5,446	5,453	64	84	40

a From 1995, DSD data re-classified beverage from steel to composites.

b CEC data allocated composites between the main materials according to their market share give in 1995 by the *Umweltsbundesamt* (German Environment Agency) (55% paper, 39% tinfoil steel, 3% plastics, 3% aluminium).

Sources: DSD (1998; 1999a), CEC (2000a: 135)

Table 5.7 UK Recovery and Recycling 2001 (thousands tonnes)

	<i>DETR estimates</i>	<i>Reprocessing capacity</i>	<i>Capacity required</i>			<i>Balance</i>
	2001	2001	50% recovery	15% recycling	25% recycling	25% recycling
<i>Recycling</i>						
Paper	4,308	1,921	2,154	646	1,077	844
Glass	2,200	730	1,100	330	550	180
Aluminium	111	53	56	17	28	25
Steel	735	235	368	110	184	51
Plastics	1,912	212	956	287	478	-266
Wood	1,300	350	650	195	325	25
Total packaging	10,566	3,501	5,284	1,585	2,642	859
25% recycling					2,642	
EfW		726				
Total recovery		4,227	5,284			
Shortfall		1,057				

Source: updated from DETR (1999a: 21)

According to ongoing reports in *Materials Recycling Week*, the price of both commodities are recovering after prolonged slumps caused by plentiful supplies of raw materials on the international market. The major obstacle for EfW, in the opinion of the DETR, is public resistance to the location of incineration plants near their communities, the so-called 'Not In My Back Yard' (NIMBY) effect (see also Goldman, 1996; Lake, 1996; Elliot, 1998; Rabl *et al.*, 1998)¹². This is particularly problematic considering the prominence of EfW in Britain's latest sustainable waste strategy, *Less Waste, More Value* (DETR, 1999b). At the time of writing, three new EfW plants are under construction and eight more are awaiting planning permission (DETR, 1999a).

However, the greatest challenges are faced by the plastics reprocessing sector, where the projected annual deficit is expected to be 43,000 tonnes against the 15% minimum and 266,000 tonnes against the overall 25% target (Bailey, 1999b; 1999d; DETR, 1999a). This has occurred despite the existence of apparently healthy competition between the 82 plastics reprocessors currently registered with the Environment Agency. The DETR report notes, however, that most major manufacturers of plastics have curtailed or entirely severed their involvement in recycling and chosen instead to concentrate on producing plastics from virgin materials (DETR, 1999a). The problem appears to be that plastics recycling is simply unprofitable, either because recycling processes are prohibitively expensive (Beynon, 1993; Bailey, 1999b; 1999d), or because collecting viable amounts of lightweight plastics waste incurs very high transport costs. Competition in the plastics sector has therefore been limited to niche markets despite the incentives provided by PRN subsidies (MRW, 1999d; 1999e). In response, the government has taken little direct action but has repeatedly emphasised that rapid expansion in plastics recycling is essential if Britain is to meet its recycling targets (DETR, 1998a; 1999a).

In fact, the problems of the plastics sector highlights one of the main problems of pursuing environmental objectives using market-led regulation. In this particular case, the British government has struggled to convince individual companies of the need to develop strategic plans for increasing the recovery and reprocessing of waste materials. However, such planning is only likely to occur where market operators perceive it will yield significant financial benefits. In simple terms, if there is little prospect of plastics

¹² Local authority waste planners expressed similar opinions during the MEL Research conference, *Strategic and Local Planning for Waste* in 1998 (MEL Research, 1998).

recycling becoming viable, there is little reward for long-term expansion. In Germany, this conflict was circumvented by the unequivocal requirements imposed by the Ordinance and the fact that the Dual System has always operated as a regulatory device rather than a 'free' market instrument. Having flirted unsuccessfully with a more voluntary approach prior to 1993, the government enforced its agenda by sanctioning increases in Green Dot charges. Once the DSD had achieved financial stability, it was able to co-ordinate its collection and investment programme accordingly. However, it is difficult to see Britain's 'free' reprocessing market undertaking such actions without clearer financial incentives. Indeed, rather than trying to exert pressure on reprocessors, some compliance schemes have simply agreed reprocessing contracts with overseas reprocessors¹³. Arguably this move may not even be related to the reprocessing-capacity problem, as there are no mechanisms preventing British compliance schemes from using foreign reprocessors if they are cheaper than their domestic counterparts, even if capacity is available. It remains to be seen whether the British government will eventually lose patience and intervene more forcibly to counteract the reprocessing sector's apparent lack of urgency.

5.4.3 *The Collection of Waste Packaging*

The second factor determining the efficacy of the British and German PWMSs is their ability to develop comprehensive networks for the collection, sorting and transfer of waste for reprocessing. In many respects, there is little to say about the German model, as the DSD's main *raison d'être* is to co-ordinate and finance the collection of post-consumer packaging waste. In 1998 it spent the equivalent of £1.4 billion on purchasing recycling services (93.7% of its turnover), 73% of which was invested in kerbside or mixed collection schemes (DSD, 1999a). Of the 537 sub-contracts the DSD operates, 104 are with local authorities, 76 are with private companies in conjunction with local authorities and the remainder are private company contracts. Against this, the DSD acknowledges that only 5.6 million tonnes of the 6.1 million tonnes of packaging waste collected in 1997 were sufficiently free from impurities that they could be recycled (DSD, 1999b). Germany's recycling rates (Table 5.6) nonetheless attest to the DSD's successes in developing a comprehensive and effective network for the collection of post-consumer packaging waste.

¹³ In the most extreme example to date, a compliance scheme took an Environment Agency delegation to China to approve a plastics reprocessing site which would significantly reduce its costs for handling this material. Other compliance schemes are reported to be considering similar moves (MRW, 1999c).

Table 5.8 Household Waste Collection necessary to fulfil the UK Regulations (thousands tonnes)

	<i>Packaging flowing into waste streams</i>		<i>Collection required for 50% recovery</i>	<i>Household collection required^a</i>
	Commercial-Industrial	Household		
Paper	3,500	500	2,000	0
Glass	350	1,850	1,100	750
Aluminium	5	105	54	50
Steel	177	573	368	191
Plastics	600	1,100	850	250
TOTAL	4,632	4,128	4,372	1,241

^a Negative figures for paper are counted as zero to factor out cross-subsidisation.

Adapted from: DETR (1999a: 18)

Although the original aim of the UK Regulations was to concentrate on the recovery of industrial-commercial waste, it was always recognised that post-consumer waste may form an important part of the overall collection effort (DETR, 1998a; 1998c; 1999b). Whilst the DETR has quantified the split between the two (Table 5.8), these figures assume that all commercial-industrial waste will be recovered. They are therefore a conservative estimate of the likely final requirement for household waste recovery. In order to examine the development of post-consumer waste collection in Britain, businesses responding to the reprocessor survey were asked to provide details of their involvement in waste collection and, in particular, their use of PRN revenue to develop partnerships with local authorities and private sector organisations. However, this analysis was hampered by the imprecise information provided by some reproducers. For example, although only 15 of the 48 respondents reported having partnerships with local authorities - suggesting low interaction between the two groups - one reprocessor claimed 'to be working with' (in an unspecified manner) over 300 councils. Moreover, most respondents were unwilling to give specific details of partnership agreements, presumably because the information was commercially sensitive. The survey results were therefore inconclusive. The DETR review was less equivocal, however, and impressed the need for reproducers to improve both the quantity and quality of household waste collection (DETR, 1999a). Yet the uneven power held by the

accredited reprocessors and local authority waste collection authorities appears to be a major obstacle to co-operation between the two sectors. When the PRN system was devised, local authorities were denied direct access to PRN revenue, making them reliant on reprocessors for additional funding (ENDS, 1996). Whilst it was envisaged that these transactions would naturally occur in response to the UK's recycling targets, this presumption overlooked the fact that individual reprocessors are not *legally* obliged to invest in waste collection. Reprocessors which have sufficient materials to keep their existing plants operating to capacity may therefore have little incentive to hand over PRN revenue to local authorities.

One compliance scheme outlined the problem and the steps it was taking to rectify it:

The main failing of the PRN system is that collection and sorting is not subsidised. As a result it is not helping the UK meet its targets and is not benefitting local authorities. The Company aims to overcome this problem by using the aggregated tonnage collected from local authorities for PRN acquisition and then dealing only with those reprocessors who would be willing to repay an element to be passed down the chain to local authorities.

In response to this situation, the government has introduced new requirements which require reprocessors to report the proportion of PRN revenue spent annually on waste collection and the development of end markets (DETR, 1998a). However, the government stopped short of prescribing minimum levels of investment in these areas (Institute of Wastes Management, 1999). The DETR review also highlighted the importance of public education and participation to the expansion of waste collection. Whilst the Regulations stipulate that compliance schemes should develop public awareness programmes, the review conceded that: 'it is not clear that these are necessarily policies with real impact' (DETR 1998a: 41). In response, the DETR has made reviews of expenditure on public education initiatives a mandatory part of compliance scheme reporting.

The British government has therefore been forced to re-regulate the recycling industry in several areas in order to prevent 'market failures' undermining the attainment of its policy objectives. In many ways, these policy iterations are analogous to the learning process undergone in the early years of the Packaging Ordinance. However, control over the expansion of waste collection for recycling remains largely out of the government's hands though, should the existing measures prove inadequate, it may be forced to intervene more stridently. In fact, two options were considered in the

government's *Less Waste More Value* consultation paper, the introduction of separate recycling targets for industrial and domestic packaging, and direct weight-based charging for household waste collection (DETR, 1998c). Both were subsequently dropped, however, separate targets because they added further complexity to the Regulations, and direct charging because of the disproportionate impact it would have on less affluent households (ACP, 1998).

By contrast, the Germany government's legalistic approach to packaging waste policy has enjoyed more obvious success. Not only have waste deposit facilities at retail outlets and the Green Dot brought recycling directly to the public's attention, the DSD has also staged a series of high profile recycling awareness days. During its 1997 event, over 300 recycling facilities opened to the public, attracting 830,000 visitors (MRW, 1998b). As a result of such activities, the DSD reports, 61% of Germans support the idea of the Green Dot and 95% sort their packaging waste for recycling. Whether similar transparency and success can be repeated in Britain is more debatable, as commercial confidentiality is an obvious concern for competing reprocessors and schemes. However, the fact that Germany has focused primarily on post-consumer waste collection and Britain on industrial waste must be a key factor in explaining the relative successes of the two systems.

5.4.4 Reprocessors and the PRN System

One of the main purposes of the PRN system was to create a market-based vehicle for generating investment in the reprocessing industry. Intuitively, it seems reasonable that accredited reprocessors should be happy with a system that supports their industry and provides considerable expansion opportunities. However, the structural problems in some sectors suggest this may not necessarily be the case. To explore this issue, reprocessors were asked their general opinions on the merits and problems of the PRN system. Responses were then classified according to whether generally positive, neutral or negative sentiments were expressed. Of the 46 reprocessors answering this question, 12 held positive views of the PRN system, 18 were negative, and 16 wanted to see it develop further before passing final judgement. Table 5.9 summarising the opinions of each materials sector shows that, on balance, there is a greater emphasis on the negative aspects of the PRN system than might have been expected. Whilst it is difficult to generalise from such limited data, the aluminium, steel, glass and EfW

sectors expressed the greatest overall satisfaction with the PRN system, whilst paper and plastics reprocessors were noticeably less content. There is also the argument that if individual reprocessors (in whatever sector) are receiving PRN revenue which has the effect of making previously marginal operations more commercially attractive, there is no reason for them to protest. If, on the other hand, PRN revenue has failed to make some sectors profitable, their disenchantment is more understandable. As one compliance scheme put it:

The glass, aluminium and steel areas have been a success simply because reprocessors have not needed to invest one jot in new capacity.

Table 5.9 Reprocessor Attitudes towards PRN System (% of companies)

	<i>Number of respondents</i>	<i>Negative</i>	<i>Neutral</i>	<i>Positive</i>
Paper	13	53.8	46.2	0.0
Glass	3	0.0	33.3	66.7
Aluminium	2	0.0	50.0	50.0
Steel	4	0.0	25.0	75.0
Plastics	17	58.8	23.5	17.7
Incineration	7	14.2	42.9	42.9

The unease about the efficacy of the PRN system is also reflected in the policy debate which accompanied the scheme's introduction and which peaked in the 1998 review of the Regulations (ACP, 1998; DETR, 1998a). The debate broadly divided into those sectors of the packaging chain which supported the existing system - generally speaking, the reprocessing industry - and those packaging producers and compliance schemes that were convinced it was being manipulated. The disagreements centred on two fundamental issues, the PRN system's ability to finance the expansion of the recycling industry and the role of reprocessors in the management of PRNs and their associated revenue.

Stabilising markets and funding expansion

As noted previously, the satisfaction with the PRN system as a method of generating investment revenue varies greatly between reprocessing sectors. Many respondents in fact viewed the PRN mechanism as extremely effective:

All PRN revenue is used to promote recycling in the UK either by supporting price, or developing infrastructure or promoting awareness campaigns. If there is a surplus of recycle the revenue can be used to support prices, conversely if there is a surplus of manufactured recycled end product then the revenue can be used to support lower sales prices to stimulate end markets. (aluminium)

The revenue from PRNs enables us to buy waste which would not otherwise be viable commercially to pay for ... and to help finance expansion schemes. (plastics)

It is our view that the current PRN situation is such that it provides economic justification for the installation of Energy from Waste facilities without which such projects would not proceed. (incineration)

The PRN system is functioning well and will continue to develop. No major changes should be made to the overall structure of the system at the present time. (steel)

The submissions from these reprocessors indicate that their perceptions of the PRN system generally correspond with those held by the Environment Agency when the scheme was conceived. Other respondents were more pessimistic about its stability and the investment conditions being created:

It is difficult to invest in new facilities when long term guarantee of price (or demand) for PRNs cannot be given. (paper/plastics)

The PRN system has at present collapsed. Unless we get stability in the market place next year ... we will have to stop using recyclable material. (plastics)

The PRN system as it stands today is unsustainable as the surplus available is undermining the price structure and forcing the raw material price down. I have given you the basis of our problems and concerns [complexity, lack of enforcement, loss of money through PRN sales] ... but if I continued, I am sure I could fill ten pages with the problems and impossibilities of the scheme. (paper)

Another plastics reprocessor gave a clearer indication of its problems. The company claimed that the cost of reprocessing one tonne of material, including purchase, delivery and reprocessing, was £270 per tonne. As the PRN value of plastics was £140 per tonne at the time of correspondence, the company averred that it was unable to make a profit even when produce sales were taken into account.

The efficacy of the PRN system as an investment mechanism in fact seems to be impeded by two factors, the variability of PRN prices on the free market and the uncertainties created by the legislative regime. Whilst it was hoped that PRN prices would respond automatically to EU recycling targets and production efficiencies, market prices have also been strongly influenced by other supply and demand factors, many of which are international in character and beyond the government's control. This has been particularly the case for paper and glass. The simple relationship between legislative standards and market actions envisaged by the government has therefore proved to be an over-simplification. The situation has been exacerbated by the fact that producers and compliance schemes are required to report their compliance to the Environment Agency on an annual basis. This has made them reluctant to sign extended contracts for PRNs and denied reprocessors the finance to plan long-term investments (Bailey, 2000). This point was made by one plastics reprocessor.

There is a potential conflict of interests between the compliance schemes and the reprocessors. Compliance schemes wish to obtain PRNs at the most economic rate and to remain flexible in what is a rapidly changing system. They are therefore reluctant to enter into longer term purchasing contracts. This can potentially work against reprocessors on both fronts by failing to create the conditions for a sustainable and expanding reprocessing market.

The Management of PRNs and Revenue by Accredited Reprocessors

A range of criticisms concerning the management of PRNs was also raised by compliance schemes and producers during the 1998 review. Firstly, they argued that the scheme did not provide producers with an incentive to set up collection schemes for packaging waste, as many reprocessors are charging for PRNs even when the producer itself collected and delivered the waste. This, along with the failure of reprocessors to fund local authority schemes, were cited as major reasons for the low collection and sorting rates in Britain. One compliance scheme noted that:

An effective PRN system should have provided free proof of recycling to the originators of the waste. Those segregating waste and making it available for reprocessing should then have benefitted from the value of any Tradeable Permits. This ... notion was ignored and instead a ... system whereby reprocessors charged fees to reprocess material has sprung up in its place.

The most cost-effective option for many producers, therefore, is simply to discharge their recovery obligations by acquiring PRNs on the open market without expanding their recycling activities. Secondly, producer associations claimed that some parties were deliberately exploiting the reporting requirements of the Regulations. They suggested that PRNs were either being withheld by reprocessors until close to the annual reporting deadline or that they were being acquired by parties unconnected with the Regulations, in both cases to speculate on their value (Bailey, 1999b; 2000):

There are concerns that this market mechanism could be abused by individuals or organisations with minimal or zero obligations under the Regulations seeking to profit by trading in PRNs, e.g. by stockpiling until the compliance date is imminent. This would merely increase the cost to obligated companies, and make no contribution to the growth of recycling. This concern would not arise if the mechanism to demonstrate compliance were to be separated from the mechanism needed to inject funds into the system as is the case with most EU Member States. (producer association)

The final allegation was that reprocessors were simply using PRN revenue to furnish company profits rather than investing in recycling infrastructure.

Those sectors requiring fundamental investment such as paper, card and plastic have demonstrated a complete inability to prove where the funds have gone - and we strongly suspect that those funding flows have propped up balance sheets or P&L at a time of falling global commodity prices. (compliance scheme)

The rationale behind charging for PRNs ... was to provide a financial incentive to stimulate collection and sortation and capital to increase the UK's reprocessing facilities. However, there is no guarantee that reprocessors will use the funds in this way. Indeed one company has been offered PRNs at £35 per tonne for waste being incinerated with energy recovery and there is no intention of using the money to increase recovery rates. The injection of revenue is moving from obligated companies to enhance reprocessor's profits, which is of no help to the environment. This is unfair, unjustified and totally against the spirit of shared responsibility under which the Regulations were developed. (producer association)

In fact, the extent of either practice has never been fully proven and the movement of PRN prices seems inconsistent with the existence of widespread profiteering. If this had occurred, PRN prices should have appreciated steadily and shown marked increases around the compliance deadline each January. However, all PRN prices have dropped markedly since the scheme was established (the value of plastics PRNs, for example, fell by 73.8% between 1998 and 2000) (Figure 5.7). Whilst the monthly price

Figure 5.7 PRN Price movements 1998-2000

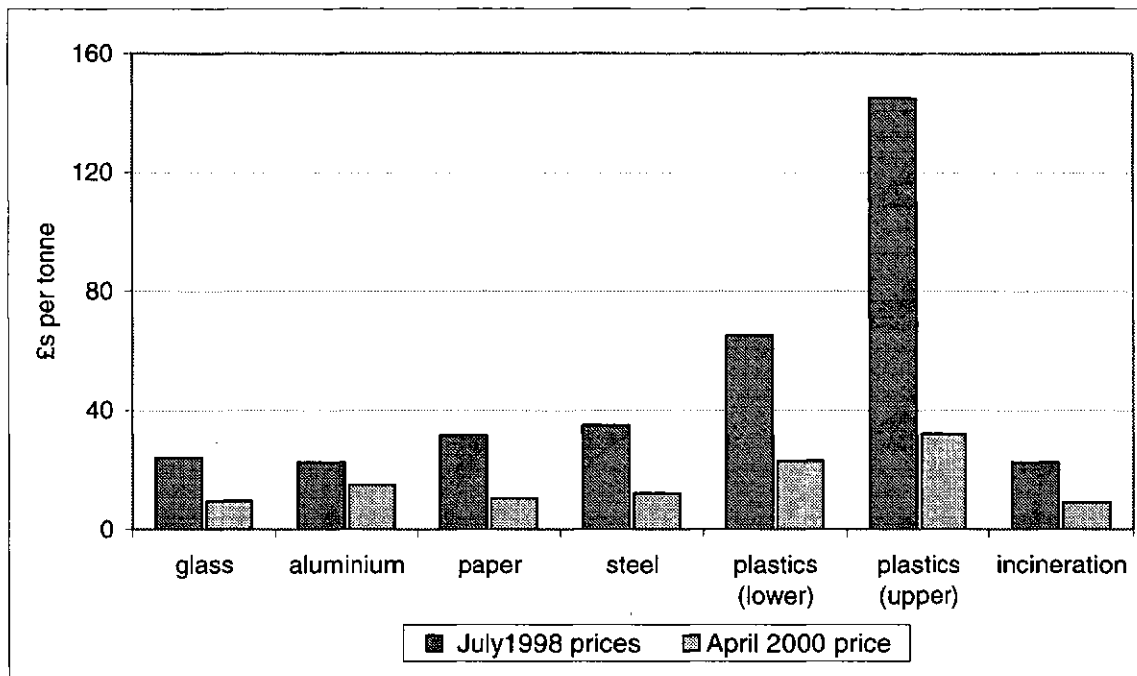
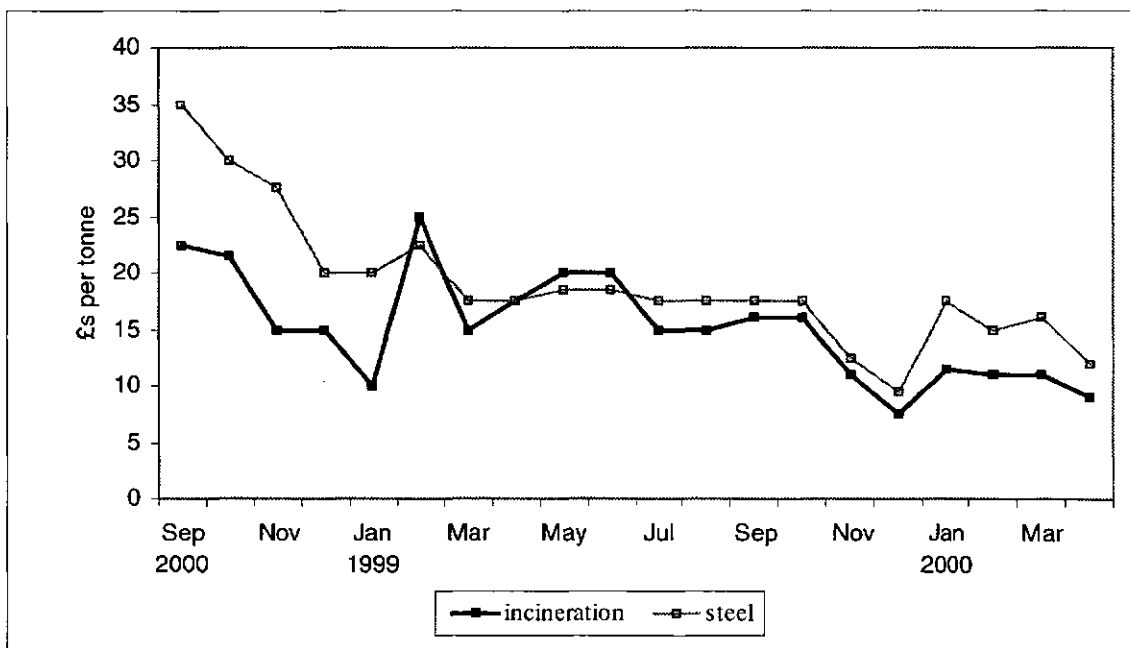


Figure 5.8 PRN Price movements, Steel and EfW Incineration



(Data Source: MRW, July 1998 - April 2000)

movements for the steel and incineration sectors, for example, (Figure 5.8) show small increases around the compliance deadlines, the downward trend has been reasonably consistent for all materials. However, the multitude of concerns about the PRN system persuaded the government that additional controls were required. In addition to the

requirement for reprocessors to report the proportion of PRN revenue diverted towards infrastructure development, the review supported the ACP's recommendation that the issue of PRNs should be restricted to those parties directly connected with the Regulations (ACP, 1998; DETR, 1998a). Aside from these restrictions, the government has supported the ACP's view that maintaining the PRN system's market-led approach is essential for the successful delivery of the EU's recovery and recycling targets (ACP, 1998).

It should be noted that the German pricing mechanism for packaging waste has also been heavily criticised in terms of its economic inefficiencies and net environmental achievements (House of Commons, 1997). Staudt (1997) and Eichstädt *et al.* (1999) both argue that recovery and recycling were already increasing before the Packaging Ordinance and therefore that effect of the Dual System was not as significant as its management claims. As evidence of this, Eichstädt *et al.* cite *Umweltsbundesamt* (Federal Environment Agency) figures showing that 1.79 million tonnes of glass were recycled in 1990 (p. 145). Staudt claims that as recycling would have reached 3.7 million tonnes per annum without the Dual System, the 'real' additional figure achieved by the DSD, 2 million tonnes, does not justify the costs involved.

Continuing the attack, Staudt also suggests that the full cost of the DSD to industry is 9 billion Deutschmarks (£3.2 billion at the time Staudt's report was compiled) rather than the official 4.4 billion Deutschmarks (£1.6 billion) reported. According to other reports, however, a large proportion of the DSD's ongoing costs are a consequence of its set up and early losses, leaving much potential for future savings (Handelsblatt, 1997). Arguably the recent reductions in Green Dot charges are a sign of this trend, as is the DSD's announcement that its waste management costs fell by 240 million Deutschmarks in 1999. Whilst a recent Commission report concluded that the German system has achieved the greatest absolute environmental benefit of four national PWMSs studied (Britain, France and the Netherlands were the others), it found that this was achieved at the highest per unit cost (CEC, 2000a). Michaelis' (1995) conclusion is that the DSD has not yet achieved economic efficiency but that it has the potential to do so. Staudt (1997) nonetheless argues that greater reliance on incineration and an integrated waste management approach would have yielded more beneficial economic, environmental and social outcomes. However, Eichstädt *et al.* (1999) produce an interesting defence of the DSD's high costs. They argue that lower charges would

make the recovery and recycling of packaging waste more economic and would therefore remove the only incentive for packaging avoidance contained in the Ordinance. Consequently, they conclude that economic efficiency on the output side (wastes) is often in conflict with environmental efficiency on the input side (packaging design) (also Brisson, 1993).

The overall question, therefore, is whether price and market-based models of environmental management are capable of producing environmentally-efficient outcomes. First, it must be recognised that as neither system is a fully-fledged market-led model but rather are *post hoc* additions to legislative intervention, many of their deficiencies relate to the fact that they must operate in sub-optimal conditions (though this was always likely to be the case). For example, the PRN system is not a market with free choice between buyers and sellers, as producers and compliance schemes are obliged to participate in order prove their compliance with the Regulations and reprocessors are constrained by the unwillingness of producers to enter into long-term contracts. As such, 'free' choice is restricted for all players in the PRN market. The only market force that might have counteracted restrictive practices, competition amongst reprocessors and compliance schemes, does not appear to have created the necessary balance. It was therefore impossible for the British government to permit these distortions to persist and, with the benefit of hindsight, the additional controls imposed were inevitable.

However, there also appear to be deep-seated structural conflicts between the priorities of the market and those of environmental efficiency. Firstly, there is the market's inability, or disinclination, to use hypothecated resources for environmental purposes. If neither market forces nor government regulation provides market players with requisite incentives to invest in environmental protection measures, investment is unlikely to occur of its own volition. Where market and policy priorities are compatible, the interests of both parties may be promoted; where there are clashes, the outcome depends on the market constraints in place. Secondly, there is the sheer complexity of market systems. Reprocessors participating in the PRN system are responding to numerous stimuli, many of which are not controlled by the Regulations. Understanding the inputs and outputs dictating market behaviour is an intricate task and may require numerous policy iterations. Whilst it would be unfair to claim that market forces have acted totally against the aims of the UK legislation, Sinclair and Fenton

(1997) argue that market mechanisms are flawed because they do not possess the natural mechanisms to overcome such impediments. A less damning assessment, echoing Turner *et al.*, (1998), would be that clear government guidance is required in the design and control of price-based environmental policy instruments if damaging disincentives are to be avoided.

5.5 Conclusions

The purpose of this chapter has been to explore the negotiation and transposition of the Packaging Directive and its implementation in two Member States. In so doing, many of the tensions associated with environmental policy-making in the EU have been highlighted. The main, though not entirely new, revelation is the extent to which Member-State interests and practices influence each level of EU decision-making. In common with most EU legislation, the Packaging Directive was the product, not of any mutually-agreed 'rational' criteria, but rather was comprised of diluted and politicised objectives intermingled with the derogations and flexibility necessary to make the policy acceptable to each Member State.

Viewed from this perspective, EU environmental policy still demonstrates many features of the lowest-common-denominator bargaining that generally characterises inter-governmental agreements. Against this, the fact that the Member States are more committed to collective action than would be the case in an entirely inter-governmental grouping enabled a more radical programme of reform to be established than some states would otherwise have accepted. This confederal element of the EU persona has enabled environmental policy to develop agreements which are more ambitious than anything hitherto established (Zito, 2000). What is less clear, however, is whether the EU's particular form of bargaining-focused policy formulation can produce adequate responses to the environmental problems confronting the EU. This point is considered further in Chapter eight.

Similar issues have arisen in the implementation of the Directive in Britain and Germany. Reflecting its desire to achieve cost-effective compliance, the UK government has adopted a relatively liberal and market-led system of packaging waste management, while successive German administrations have used constrictive policies to defend their chosen environmental principles and practices (Ramus, 1991; Lowe and Ward, 1998a). The result in Germany is a system which has met its stated

environmental objectives but has yet to achieve economic efficiency (Michaelis, 1995; Staudt, 1997). The British model, by contrast, is relatively cost efficient but is struggling to attain its environmental targets. Although the two governments have always differed in the way they prioritise and interpret environmental policy (Gee, 1997; Lowe and Ward, 1998a), price-based regulation, by its very nature, may further integrate and entrench these differences. If this occurs, then the gap between the EU's environmental 'leader' and 'laggard' states may widen and the common strand of sustainable development linking them may become harder to discern.

However, the study also revealed areas where the policy styles of the two countries are converging. The British government, whilst not revoking its 'free' market principles, has started to regulate its recycling market more earnestly. Similarly, the debate in Germany has increasingly focused on methods for improving the economic efficiency of the Ordinance, including the introduction of competitive pressures and greater financial accountability. Therefore, despite the ideological differences that initially informed each policy approach, both models have been forced to reconsolidate in response to common practical pressures. Whilst total harmonisation is unlikely, the EU policy process provides numerous opportunities for national and regional authorities to transfer knowledge and best practice. If these possibilities are properly exploited, they should prove invaluable in the development of cost-efficient and sustainable environmental management systems in the Member States.

Chapter Six

Industry's Response to the Packaging Waste Directive

6.1 Introduction

Having examined the general frameworks of packaging regulation in Britain and Germany, the next two chapters assess their impact on business waste management strategies. Chapter six compares the extent to which British and German businesses are responding to the Directive's objectives in terms of (i) the mandatory recovery and recycling standards outlined in Chapter five and (ii) the 'essential objectives' contained in Annex II of the Directive. This latter category encompasses the prevention and re-use of packaging waste as well as the development of end-use markets for recycled materials. Where appropriate, the policies are also assessed in terms of whether business actions are consistent with the generic environmental principles established in the EAPs. These include the Polluter Pays Principle, the Proximity and Precautionary Principles, the adoption of preventative rather than 'end-of-pipe' cures for environmental problems and the integration of environmental considerations into all spheres of policy and business activity. Chapter seven then explores the impact of economic instruments and, specifically, the influence of environmental charges on polluter behaviour. The principal data used in both chapters are derived from the survey of UK and German packaging producers.

Chapter six begins with a preliminary assessment of the data. The purpose of this is to identify any factors that might prejudice comparisons between the German and UK respondent groups (this procedure is recommended by Silk (1979), also Shaw and Wheeler (1994) and de Vaus (1996)). This is followed by an appraisal of business reactions to mandatory recovery and recycling targets as well as the Directive's essential objectives. The final section presents qualitative feedback from businesses on the efficacy and impact of national packaging regulations.

6.2 Profiles of the Respondent Groups

Several facts about the two respondent groups were established in Chapter four. Firstly, the number of businesses replying to the survey was greater than anticipated

during the research design. In total, 52.1% of British and 34.3% of German companies completed the questionnaire. Secondly, it was argued that the sampling process was largely successful in identifying businesses affected by national packaging legislation (see Tables 4.3 and 4.4). However, because the German database was compiled independently rather than using an official directory of obligated companies, it was important to identify any discrepancies between the two groups¹. Two preliminary filter questions were therefore used to establish whether potential respondents were (i) aware of national packaging legislation, and (ii) subject to specific legal duties as a result (see Chapter four). These data, analysed using Chi-Square tests (Table 6.1), confirmed that 96.4% of UK respondents are obligated by the Packaging Regulations and that 76.4% of German businesses are affected by the Packaging Ordinance. This therefore reduced the number of valid responses to 450 for the UK (50.0%) and 236 for Germany (26.2%).

The next stage was to compare the characteristics of the two respondent groups. These were measured in the survey in terms of (i) company turnover, (ii) number of employees, (iii) business sector within the packaging chain and (iv) main activity. It should be remembered, however, that some variations were expected as the Ordinance and Regulations distribute recycling responsibilities differently amongst the sectors of the packaging chain. Any variations between the groups should nonetheless be recognised and assessed. Chi-Square tests were again used to examine respondent profiles and revealed significant variances between the samples for all profile variables (Table 6.1). In terms of company turnover, the principal differences were caused by the higher representation of German businesses with annual earnings of either below £5 million or over £1,000 million. As the sampling procedure attempted to remove German firms with a turnover of below £5 million, their presence indicates that there were inaccuracies in the business directory used to compile the sample. However, these respondents were not removed from the sample because the Ordinance does not apply the £5 million minimum turnover threshold used in the Regulations (see Chapter five). Moreover, as both samples are dominated by businesses in the £5-49 million turnover category, the variances were considered to be within acceptable limits.

¹ Though the German sampling frame was compiled following extensive analysis of the Packaging Ordinance and other relevant literature (including other academic and professional commentaries).

Table 6.1 Analysis of Respondent Characteristics

	<i>Germany</i>	<i>UK</i>	<i>Chi-Square (χ^2) and sig.</i>
<i>No. of business aware of national packaging legislation</i>	288	463	16.923 0.000
<i>No. of businesses affected by packaging legislation</i>	236	450	67.163 0.000
<i>Turnover (£ million per annum)</i>			
0-4.99	22	3	
5-49	129	272	
50-99	27	65	
100-499	24	63	
500-999	8	16	
1,000+	19	12	
<i>Total</i>	229	431	45.278 0.000
<i>Number of employees</i>			
0-49	10	52	
50-99	47	75	
100-499	98	192	
500-999	21	60	
1,000-4,999	13	45	
5,000+	8	24	
<i>Total</i>	197	448	13.843 0.017
<i>Main business activity</i>			
Domestic trade	161	230	
Import	6	29	
Export	18	28	
More than one category	24	150	
<i>Total</i>	209	437	45.961 0.000
<i>Packaging chain sector</i>			
Raw materials producer	2	8	
Packaging manufacturer	4	39	
Product manufacturer	141	229	
Wholesaler	21	33	
Retailer	22	35	
More than one category	42	93	
<i>Total</i>	232	437	16.663 0.005

A similar pattern emerged with respect to the average number of employees per company. In this case, there was a concentration in both countries of respondents with 100-499 employees and a higher representation of German businesses with 50-99

employees. The variance can again be explained by differences between the Ordinance and the UK Regulations, as can the greater number of product manufacturers (packer-fillers) in the German sample. Whilst the Ordinance focuses the majority of recycling responsibilities on manufacturers and retailers, the Regulations spread obligations across all sectors of the packaging chain. Thus, a degree of imbalance was to be expected. The variations in main business activity are less readily explained except in terms of the chance self-selection of respondents. Here, a greater proportion of German respondents classified themselves as dealing exclusively in domestic trade or exports and a larger share of British firms companies were engaged in more than one category of commerce. Though this may reflect the broader trade profiles of German and UK industry, it must be remembered that circumstances prevented the compilation of an ideal sampling frame for Germany. Working with limited and imperfect data is a frequent hazard in scientific research, particularly in the social sciences (Sarantakos, 1993). However, whilst there were clear differences between the profiles of the two respondent groups, the majority can be explained by legislative factors. Their influence on business waste management practices was nonetheless monitored throughout the data analysis.

6.3 Waste Management Actions in the UK and Germany

6.3.1 Introduction

When planning the survey strategy, consideration was given to the merits of asking respondent businesses if they intended to, or were capable of, achieving the statutory targets set by the Packaging Directive. This approach was seen as potentially perilous, however, first, because it might have seriously reduced the response rate and, second, answers to this question would be likely to contain a strong element of social desirability bias². In short, it seemed improbable that companies would confess to an illegal act even under the protection of respondent confidentiality. It was therefore decided that an implicit assumption should be made that all affected businesses would achieve their statutory obligations. That said, there is evidence that such a presumption might be over-optimistic. Both the German and British governments have been forced to embark on major programmes to pursue free-riders (Michaelis, 1995; Perchards,

²Social desirability is the phenomena whereby survey respondents provide answers they believe will reflect them in a good light. The issues of sensitive questions and social desirability responses are extensively discussed in Moser and Kalton (1971) and Czaja and Blair (1996).

1998; DETR, 1998a; Bailey, 1999c). Moreover, there were sizeable discrepancies between DETR and business estimates of the total packaging produced in the UK in 1997-8 (DETR, 1999a). The total packaging production reported by UK industry fell from 12.5 to 7.7 million tonnes between 1997 and 1998, compared with DETR estimates of 11.8 million and 10.2 million tonnes for the same years. Although the DETR figures are acknowledged to be best estimates, these discrepancies raise doubts as to the honesty of some industry reporting.

The problem was partially circumvented by asking respondents to provide details of their involvement in the physical collection of packaging waste. Whilst this does not directly measure the proportion of firms achieving legal compliance, it does quantify their active participation in waste reclamation. As neither set of regulations currently compels businesses to recover their packaging waste physically (see Chapter five), evidence that they are doing so would be a reasonable, if imperfect, indication of their commitment to recycling. However, this does not alter the fact that potentially relevant information had to be foregone in order to increase the survey response rate. Business responses were therefore measured against the four main requirements of the Directive and the waste management hierarchy (CEC, 1996d; Wilson, 1996); (i) the collection of waste, (ii) the prevention or source reduction of packaging waste, (iii) the use of re-usable packaging and (iv) the purchase of packaging made from recycled materials. For each variable, business response was measured in terms of the proportion of companies engaged in each activity and their level of involvement measured as a percentage of total packaging production or use.

6.3.2 Characteristics of the Survey Variables

The first stages in the data analysis were to review the basic characteristics of the data and to establish the most appropriate statistical procedures. Such preliminaries (recommended by Ebdon (1985), Griffith and Amrhein (1991) and Shaw and Wheeler (1994)) involve the production of descriptive statistics and histograms for each variable. These procedures revealed that few of the ordinal or interval variables were normally distributed but were instead positively skewed. This meant that the majority

of statistical analysis was conducted using non-parametric techniques (Silk, 1979; Norcliffe, 1982; Clegg, 1990)³.

6.3.3 Packaging Waste Collection

Two measures were used to examine the collection of packaging waste by respondent companies, the reclamation of waste materials at business sites and the physical recovery of packaging back through the supply chain. The purpose of making this distinction was to determine the precise nature of industry involvement in packaging recovery. Fenton and Sinclair (1996) stress that producers of packaging should assume responsibility for the full environmental impact caused by products they introduce to the market-place, a code they term environmental stewardship (also Lerner, 1993). Thus, environmental stewardship involves a duty of care throughout the production, sales, use and disposal of products. Although it is not always environmentally or economically effective for businesses to recover packaging waste from customers, especially where this involves substantial transportation, such actions are indicative of producers' willingness to participate in active pollution stewardship (Hill *et al.*, 1994; Powell *et al.*, 1996; Barrett and Lawlor, 1997). One example of 'enforced' stewardship in Germany, reported in Chapter five, is the Packaging Ordinance's requirement that distributors and retailers provide packaging disposal points at retail outlets (DSD, 1998). As no corresponding obligation exists in the UK Regulations, the expectation was that more German businesses would be engaged in waste collection.

Table 6.2 shows Chi-Square analyses for the collection of packaging waste from producer sites and customers. It demonstrates that though the majority of respondent companies in both countries collect production waste, 85.3% of German respondents have recovery schemes compared with 64.9% of Britain firms (Chi-Square (χ^2) = 30.892, significance = 0.000). However, only a minority of companies in either country collect packaging waste from their customers (18.4% in Germany, 13.4% in Britain, χ^2 = 2.820, significance = 0.093). Therefore, on the evidence available, there is no appreciable difference between the attitudes of German and UK businesses towards post-consumer waste collection. Considering that most sales packaging

³ The majority of the data in the set were either nominal or ordinal scale measurements of respondent characteristics or attitudes. Only percentage waste management targets and financial costs to producers were measured as interval-ratio data. The minimum acceptance criterion for all statistical tests was set at the 95% confidence limit.

remains with goods until the point of final consumption, this result is not surprising (Levy, 1993). It also reflects the fact that most waste collection is organised by the DSD in Germany or local collection authorities in Britain. This suggestion was further corroborated by the finding that German wholesalers and retailers - the only sectors obliged to provide collection facilities - were more heavily involved in post-consumer waste recovery ($N = 221$, $\chi^2 = 21.351$, $df = 5$, $p = 0.001$). Unsurprisingly, considering the broader split of sector responsibilities in the UK Regulations and its lack of provisions *forcing* retailers to provide waste disposal facilities, no corresponding relationships were found in the UK data ($N = 431$, $\chi^2 = 3.990$, $df = 5$, $p = 0.551$).

Table 6.2 British and German Packaging Collection Plans

	<i>Germany</i>	<i>UK</i>	<i>Chi-Square (χ^2) and sig.</i>
No. of businesses collecting packaging waste from business premises	193	279	
% of total respondent group	85.4	64.5	30.892 0.000
No. of businesses recovering packaging waste from customers	41	58	
% of total respondent group	18.4	13.4	2.820 0.093
<i>Packaging collection rates 1997</i>			<i>Mann-Whitney U and 2-tail sig.</i>
Number of respondents	166	330	
Mean collection rate (%)	40.3	21.1	
Median collection rate (%)	27.5	5.0	
Mean rank	303.4	220.9	
Mean rank and sum of ranks	50369.5	72886.5	18271.5 0.000

In fact, the only company characteristic with a significant bearing on waste collection in the UK was the number of employees per company, where smaller businesses were less likely to collect packaging at their own premises than larger companies ($N = 428$, $\chi^2 = 14.372$, $df = 5$, $p = 0.031$). This was presumably because waste collection is a greater drain on resources for smaller companies. However, the division between

SMEs and larger companies was not significant in Germany ($N = 201$, $\chi^2 = 5.204$, $df = 5$, $p = 0.392$), again because the collection of sales packaging is organised by the Dual System. As the DSD only deals with sales packaging, this also explains why more German businesses collect their post-production packaging waste (see Chapter five). However, whilst a greater proportion of German businesses are recovering their packaging waste, both groups have apparently avoided the complex and costly task of reclaiming waste through the supply chain. The fact that British companies do not receive PRNs free of charge for 'self-collected' waste has undoubtedly contributed to their antipathy (see Chapter five). Also, the exclusion of non-sales packaging from the Dual System - but its inclusion in business' overall recycling targets - may well have encouraged higher recovery rates in Germany.

The next stage was to compare the average packaging collection rates of the two respondent groups. In the survey, this was measured as the percentage of total packaging produced or used by respondents in 1997 (Table 6.2). The results again indicate that German businesses are collecting a significantly higher proportion of their total packaging, a mean of 40.3% (standard error of mean (SE Mean) = 2.81), compared with 21.1% in the UK (SE Mean = 1.55). The median packaging collection rates were 27.5% for Germany and 5% for the UK. Mann-Whitney tests confirmed that these differences were over 99.9% significant.

Kruskal-Wallis H tests were then used to analyse the influence of company characteristics on waste collection rates. The only significant relationship revealed was that smaller businesses in the UK generally recover less packaging ($\chi^2 = 11.445$, $df = 5$, significance = 0.043). This stage of the survey has therefore revealed three major points. First, significantly more German firms are involved in collecting packaging waste, though both groups have generally ignored post-consumer waste recovery. Second, German companies are, on average, recovering a larger proportion of their packaging waste. Third, this commitment seems to extend to all sectors of German industry. It is particularly noteworthy that the German SME sector has taken on greater waste management responsibilities. As SMEs usually find it more difficult to devote resources to non-core activities, this suggests that active waste management is becoming firmly embedded in the German economy.

6.3.4 Source Reduction

The next stage of the analysis was to examine business behaviour in relation to the Directive's primary essential objective, the prevention or source reduction of packaging waste. In order to assess this, survey respondents were asked to provide details of any programmes they had to reduce the amount of packaging produced by their companies. 57.1% of German respondents had reduction plans by 1998 compared with 12.8% of British businesses (Table 6.3). This corroborates the findings from a survey by UK trading standards authorities, where just 9% of products were adjudged to make efficient use of packaging materials. Of 105 products tested, 15 contained twice the amount of packaging required to contain the product adequately (ENDS, 1999).

The methods used by companies to reduce packaging were also explored in the survey (Table 6.3). Two options were considered; reductions in the total amount of packaging consumed by respondent firms, and the 'light-weighting' of packaging (the technique of reducing the weight of packaging per product unit). The importance of this distinction is that it highlights the extent to which businesses are placing *absolute* or *relative* limits on their use of packaging and therefore relates back to the notions of weak and strong sustainability outlined in Chapter two. Weak sustainability, it will be remembered, generally supports the substitution of natural capital for human capital provided production and consumption processes are made less environmentally damaging. However, strong sustainability places a stronger emphasis on acknowledging absolute carrying capacities which human society must operate within.

Huppes *et al.* (1992) argue that the majority of environmental legislation imposes relative improvement or prohibition standards because they specify minimum per-unit quality standards. This is not true in all cases; the Montreal Protocol, for example, imposed a complete ban on the production of CFC gases. Similarly, participants at the Rio Conference committed to reducing their greenhouse gas emissions to 1990 levels by the year 2000 (DoE, 1996a; Lowe and Ward, 1998a; Oberthür, 1999). Furthermore, not all economic instruments operate in a relativist manner. Tradeable permits, for instance, set absolute ceilings on prescribed pollutants (Pearce *et al.*, 1995). Relative quality standards nonetheless remain the predominant form of policy intervention. Supporters of strong sustainability argue that this approach has severe deficiencies where quality standards contain limited reference to the environment's resilience to resource depletion (Forrester, 1999). Relative reductions in packaging consumption

might therefore be taken to be a policy expression of the weak sustainability approach to environmental management, whilst absolute targets are more analogous to the concept of strong sustainability.

Table 6.3 British and German Packaging Reduction Plans

	<i>Germany</i>	<i>UK</i>	<i>Chi-Square (χ^2) and sig.</i>
No. of businesses with packaging reduction programmes	125 (57.1%)	55 (12.8%)	142.429 0.000
<i>Type of reduction programme</i>			
None	75	362	
Reduction in total packaging use	67	35	
'Light-weighting' of packaging	35	18	
Combination of both methods	15	11	135.406 0.000
<i>Planned packaging reduction percentages 1997-2001</i>			<i>Mann-Whitney U and 2-tail sig.</i>
Number of respondents	195	401	
Mean reduction rate (%)	10.8	2.0	
Median reduction rate (%)	5.0	0.0	
Mean rank	387.3	255.3	
Mean rank and sum of ranks	75522.5	102383.5	21782.5 0.000

Although 85% of UK respondent businesses and 39.1% in Germany had adopted neither strategy, 34.9% of German respondents have committed to reductions in their total packaging consumption. This compares with 8.2% in Britain. A further 18.2% of German companies and 4.2% of British firms have begun light-weighting programmes. The percentage of businesses adopting a combination of the two approaches was 7.8% in Germany and 2.6% in Britain. The main explanation for these differences appears to be the way in which the German and British charging systems for packaging waste operate⁴. Because Green Dot fees are charged for all packaging regardless of whether it is recycled, they create a stronger incentive for German industry to reduce excess packaging. By contrast, British producers are only required to pay for PRNs according to the recovery and recycling targets contained in the Regulations (see Chapter five).

⁴ Though a code of practice for packaging design was also published in the UK (INCPEN, 1998).

Furthermore, the fact that recycling charges are substantially higher in Germany makes it more difficult for businesses to recoup their environmental costs from consumers. Greater changes in corporate practice are therefore encouraged. Though it would be ambitious to claim that either system has achieved a strong sustainability outcome, the German model appears to be more precautionary and is closer to defining overall limits on the production of packaging waste.

The mean packaging reduction rates in each country (Table 6.3) further corroborated this conclusion. German businesses with reduction programmes predicted they would reduce their packaging use by 10.75% between 1997 and 2001 (SE Mean = 1.10, median = 5.0%). This compares with a mean of 1.96% for the UK (SE Mean = 0.32, median = 0.0%). Chi-Square and Kruskal-Wallis tests were again used to test the impact of company profile characteristics on the packaging reduction variables but no significant relationships were found. Again it would be premature to conclude that the *majority* of businesses in either country are reducing packaging consumption. However, the notions of preventative action and quantitative limits do seem to have gained a firmer foothold in Germany than the UK.

6.3.5 Packaging Re-use

After the reduction of packaging waste, the re-use of materials is often, though not universally, considered the least environmentally deleterious method of waste management (Barrett and Lawlor, 1997; Gray, 1997). Whilst a number of environmental leader states, including Germany and Denmark, have made re-use provisions a central part of their PWMSs (see Chapter five), the flexible approach of the UK Regulations sought to avoid such methodologically-prescriptive legislation. It was therefore expected that the survey would uncover sizeable differences between each respondent group's commitment to packaging re-use.

Table 6.4 summarises the number of companies in each country with re-use schemes in operation. In total, 23.9% of UK companies had re-use plans, compared with 52.5% in Germany. However, though the difference between the mean packaging re-use targets set by the two groups was still statistically significant, it was not as extreme as for reduction. The mean re-use rate in Germany is predicted to increase between 1997 and 2001 from 24.3% to 32.1% (SE Mean = 2.33), whilst the UK re-use rate is anticipated

to rise from 10.5% to 15.8% in the same period (SE Mean = 1.15). Though this indicates that both respondent groups are exploring the potential of re-use systems⁵, the difference in participation rates can be explained by the fact that re-fill systems are a formal part of the Ordinance but not the UK Regulations (Eichstädt *et al.*, 1999).

Table 6.4 Packaging Re-use by UK and German Companies

	<i>Germany</i>	<i>UK</i>	<i>Chi-Square (χ^2) and sig.</i>
No. of businesses with packaging re-use programmes	116 (52.5%)	103 (23.8%)	53.535 0.000
<i>Packaging re-use 1997</i>			<i>Mann-Whitney U and sig.</i>
Number of respondents	166	321	
Mean re-use rate (%)	24.4	10.5	
Median re-use rate (%)	10.0	0.0	
Mean rank	298.9	215.6	
Mean rank and sum of ranks	49614.5	69213.5	17532.5 0.000
<i>Planned packaging re-use 2001</i>			
Number of respondents	155	303	
Mean re-use rate (%)	32.1	15.8	
Median re-use rate (%)	20.0	2.5	
Mean rank	277.7	204.9	
Mean rank and sum of ranks	43037.5	62073.5	16017.5 0.000

However, one important qualification is necessary here. Packaging made from wood, including wooden pallets, is included in the UK Regulations for the first time in 2000⁶. As most pallets are already managed using closed-loop re-use systems (for example, the GKN Chep system), this has undoubtedly inflated the UK re-use figures. It is unlikely that wood has had the same effect on German re-use rates as wood is rarely used in sales packaging (the main focus of the Ordinance) and therefore a large proportion of the German figures can be attributed to the re-fill targets for beverage

⁵ Examples include the introduction of re-usable trays for the transport of fresh foods by Tesco's and the use of 'tote bin' containers for small products by Makro UK. In both cases, these goods were previously transported in single-trip cardboard boxes.

⁶ The Environment Agency confirmed the inclusion of pallets in the Regulations during a personal correspondence.

containers. Consequently, it seems that German businesses are appreciably more advanced in the development of re-use systems than their UK counterparts. Again the impact of company profile, particularly business size, was minimal. However, in both cases, exporters were more likely to have re-use systems in operation. In light of the high cost of back-loading waste packaging, this is a surprising result, though again the influence of wooden pallets on these figures should not be discounted.

6.3.6 The Purchase of Recycled Packaging

The development of end-use markets is the obvious and logical outcome of any recycling system. Recycling in fact achieves few environmental benefits if reclaimed materials are not re-utilised to conserve energy and natural resources. Indeed, recycling usually involves extensive transportation networks and significant detrimental environmental impacts. Recycling can therefore only be justified where there are healthy markets for the end produce. Barrett *et al.* (1997: 113) provide an illustration of the energy savings gained from recycling packaging materials (see Table 6.5). The DETR reviews of the UK Regulations have therefore focussed on the development of end markets, whilst the DSD is founded almost entirely on the notion of 'closing the recycling loop' (DETR, 1998a; 1999a; DSD, 1998)⁷.

Table 6.5 External benefits of Recycling, Energy Saved

<i>Material</i>	<i>Value of energy saving per tonne of recycled material</i> £
Aluminium	186
Glass	2
Paper	24
Tinplate (steel)	16
Plastics (average)	148

Barrett *et al.* (1997: 113)

⁷ Closing the loop refers to systems whereby materials are extracted from waste chains then reprocessed and re-utilised, thereby diminishing the need for new resource depletion. A variant of this is down-cycling, a term used to denote materials reprocessed for secondary uses, such as the manufacture of park benches from recycled plastics. The German government generally frowns upon this practice on the grounds that it only delays the disposal of re-usable resources.

Whilst all sectors of society can contribute to the development of end-use markets, both governments have recognised that industry's purchasing behaviour is a key component of this loop. Its involvement was assessed in the survey by asking respondents to quantify the percentage of their total packaging produced using recycled materials (Table 6.6). The results showed that 53.2% of German respondents utilised recycled packaging in 1997 compared with 24.9% of British firms. Similarly, the percentage of recycled packaging used by respondent companies in Germany is predicted to increase from 32.3% in 1997 (SE Mean = 2.62) to 47.36% by 2001 (SE Mean = 2.92). This compares with a rise in the UK from 14.38% (SE Mean = 1.35) to 20.49% (SE Mean = 1.69). The median rates indicate an even greater gap, 30% in Germany in 1997 rising to 50% in 2001, against 2% and 10% for UK respondents.

Table 6.6 Purchase of Recycled Packaging by UK and German Companies

	<i>Germany</i>	<i>UK</i>	<i>Chi-Square (χ^2) and sig.</i>
No. of businesses purchasing recycled packaging	115 (53.2)	105 (24.9)	50.860 0.000
<i>Recycled packaging purchases 1997</i>			<i>Mann-Whitney U and sig.</i>
Number of respondents	161	318	
Mean % of total use	37.2	14.4	
Median % of total use	30.0	2.0	
Mean rank	307.4	205.9	
Mean rank and sum of ranks	49494.0	65466.0	14745.0 0.000
<i>Planned recycled packaging purchases 2001</i>			
Number of respondents	154	278	
Mean % of total use	47.4	20.5	
Median % of total use	50.0	10.0	
Mean rank	274.3	184.5	
Mean rank and sum of ranks	42243.5	51284.5	12503.5 0.000

Notwithstanding these differences, there is a clear upward trend in both sets of data, reflecting the importance of end markets to the British and German PWMSs. Whilst this is an encouraging trend both from an environmental and an economic viewpoint, it

is possible that the responses were again influenced by 'social-desirability' factors. Indeed, all the data have conceivably been exaggerated by businesses wishing to present themselves as responsible environmental stewards. Whilst there is no research technique which can entirely eliminate social-desirability distortions (May, 1997), the consistency of the data suggests that the trends uncovered were reliable even supposing the figures were slightly inflated.

Unlike the majority of other waste management indicators, the purchase of recycled packaging was influenced by several business profile factors. For example, UK importers and exporters were, on average, less likely to buy recycled packaging ($\chi^2 = 7.818$, $n = 421$, $df = 3$, significance = 0.050), whilst larger companies and product manufacturers generally used more recycled materials (Kruskall-Wallis (K-W) = 16.666, $n = 316$, $df = 5$, significance = 0.005). In Germany, firms with a higher turnover were more likely to buy recycled packaging ($\chi^2 = 16.437$, $n = 213$, $df = 5$, significance = 0.006, K-W = 21.012, $n = 159$, $df = 5$, significance = 0.001), as were German product manufacturers (1997 K-W = 9.926, $n = 159$, $df = 4$, significance = 0.042).

There is no obvious explanation why larger companies should be more inclined to buy recycled packaging than SMEs, though information asymmetry and the ability of larger businesses to extract price concessions from would-be sellers may be explanatory factors. The antipathy of UK importers and exporters towards recycled packaging is easier to account for, however. Importers, in particular, have limited visibility and control over their materials sources (though some UK retailers have been especially active in developing 'environmentally-ethical' procurement policies), whilst exporters have few incentives to embrace packaging stewardship for products which are excluded from the UK Regulations (DETR, 1997a). Finally, the willingness of manufacturers to buy recycled packaging is explained by the fact they have substantial influence over the design of packaging. Arguably the main barriers to the use of recycled packaging, however, are government hygiene and safety standards, particularly for human-food products (Wills, 1975; Stillwell *et al.*, 1991; Bickerstaffe and Barrett, 1993; Producer Responsibility Industry Group, 1994). This therefore reinforces the point made in previous chapters that though the waste hierarchy provides a general guide to policy

decisions, practical considerations can impede the development of recycling and closed-loop systems of waste management⁸.

6.4 Relationships between the Waste Management Variables

6.4.1 Correlation and Regression Analysis

The first stage in analysing the associations between waste management variables measured was to conduct Spearman's correlation tests. The results, shown in Table 6.7, highlight an interesting and surprising variation between the two groups. Significant correlations were found between all waste management variables in the UK data, but between relatively few indicators in the German set. Whilst there is no obvious reason for this, it might suggest that UK companies which have engaged in active waste management programmes (remembering this is only a minority of all UK respondents) have adopted a blanket approach, whereas those in Germany have developed more specialised strategies. However, it would still be misleading to suggest that many UK companies are overtaking their German counterparts, since the mean collection, reduction and re-use rates in Britain remain well below those in Germany⁹.

The next stage was to conduct multivariate analysis of the waste management variables. However, the strong associations in the UK data prevented the use of multiple regression, as one requirement of the technique is that independent variables must not be significantly auto-correlated (Shaw and Wheeler, 1994). Furthermore, the fact, for example, that 2001 re-used packaging rates are highly correlated with 1997 re-use rates scarcely enlightens the discussion. Conversely, multiple regression techniques did not produce any significant findings concerning the nature of German industry's waste management practices. It was therefore decided that multivariate models did not warrant inclusion, as they did not add substantively to the picture already established.

⁸ The view expressed by EUROOPEN is that 'waste management decisions need to take account of the nature and composition of waste streams ... Therefore flexibility is crucial' (EUROOPEN, 1997: 2).

⁹ Bhargava and Welford (1996) and Hutchinson (1996) classify business environmental responses on a continuum they term ROAST (Resist, Observe, Accommodate, Seize and Transcend). The majority of British business responses to packaging legislation can best be described as observance of legislation.

Table 6.7 Correlation of Waste Management Variables, UK and Germany

	<i>% reduction target</i>	<i>1997 % re-used packaging</i>	<i>2001 % re-used packaging</i>	<i>1997 % recycled packaging</i>	<i>2001 % recycled packaging</i>
<i>UK</i>					
% packaging collected	0.219**	0.488**	0.505**	0.508**	0.569**
% reduction target	-	0.175**	0.283**	0.149**	0.198**
1997 % re-used packaging	-	-	0.877**	0.526**	0.536**
2001 % re-used packaging	-	-	-	0.537**	0.578**
1997 % recycled packaging	-	-	-	-	0.942**
<i>Germany</i>					
% packaging collected	0.039	0.036	0.052	0.036	0.089
% reduction target	-	0.209	0.297**	0.228**	0.328**
1997 % re-used packaging	-	-	0.937**	0.163	0.126
2001 % re-used packaging	-	-	-	0.132	0.179*
1997 % recycled packaging	-	-	-	-	0.940**

* = significant at 95% confidence, ** = significant at 99% confidence.

6.4.2 Aggregated Waste Management Indices

In Chapter four, a number of waste management indices were developed for comparing the aggregate responses of businesses to packaging waste legislation. Bearing in mind the results from previous sections, it was virtually certain that cumulative statistics would merely underline the distinctions already identified. Before the indices are examined, however, it is useful to re-cap on their construction and logic. The *WMHA* index calculates the total number of waste management activities undertaken by individual respondents (collection from own sites and customers, reduction, re-use and purchase of recycled packaging) and assigns each a score of five points. The *WMHA₁* calculates the number of waste management activities in the same way as *WMHA*, then adds an extra point for every material included under each action. Thus, a business re-using each material covered by the Directive scores five points for its action and six additional points for the materials. Finally, the *WMHT* index adds together the percentage targets for all waste management actions (taking a mean of the 1997 and 2001 targets for re-use and the purchase of recycled packaging). Accordingly, the maximum possible *WMHT* score a company could be credited with was 400.

As predicted, the aggregated measures (Table 6.8) merely confirmed that German businesses are more actively involved in all aspects of packaging waste management than their UK counterparts. However, Labatt (1997a; 1997b) also used the consolidated indices to classify business' environmental behaviour (Chapter four, Figure 4.4). Under Labatt's taxonomy, companies scoring 0 on each index were classified as *reactive*, those scoring between 1 and the mid-point of the scale were deemed to be *accommodating*, and those scoring above the mid-point were considered to be environmentally *proactive*. A similar procedure was used for the *WMHA*, *WMHA₁* and *WMHT* indexes (Figures 6.1-6.3). The first observation for each index was that a comparable proportion of British and German respondents were accommodating national legislation by initiating some form of waste management policies other than straightforward disposal. The other major finding was that most German businesses are either accommodating or proactive but that British companies are predominantly reactive or accommodating. This contrast is particularly noticeable in the *WMHA* index, where the majority of German businesses fell within the proactive category. The main problem with Labatt's technique, however, was that it failed to differentiate between businesses at the upper and lower end of the accommodating and

proactive categories. For example, it did not distinguish between companies scoring 1 and 199 points on the *WMHT* scale. Conversely, whilst it might be possible to make the scale more sensitive by introducing more categories, this would involve subjective classifications and would not produce a defensible classification. Labatt's scales must therefore be seen as a relatively crude measure of business environmental behaviour.

Table 6.8 Consolidated Waste Management Hierarchy Indices

		<i>N</i>	<i>Mean Rank</i>	<i>Sum of ranks</i>	<i>Mann-Whitney U</i>	<i>2-tailed sig.</i>
WMHA Index	UK	436	277.28	120895.0		
	Germany	227	437.10	99221.0		
	Total	663			25629.0	0.000
WMHA ¹ Index	UK	436	260.31	113495.5		
	Germany	203	448.20	90984.5		
	Total	639			18229.5	0.000
WMHT Index	UK	427	274.82	117347.0		
	Germany	221	420.49	92929.0		
	Total	648			25969.0	0.000

Figure 6.1 Company Classifications WMHA Index

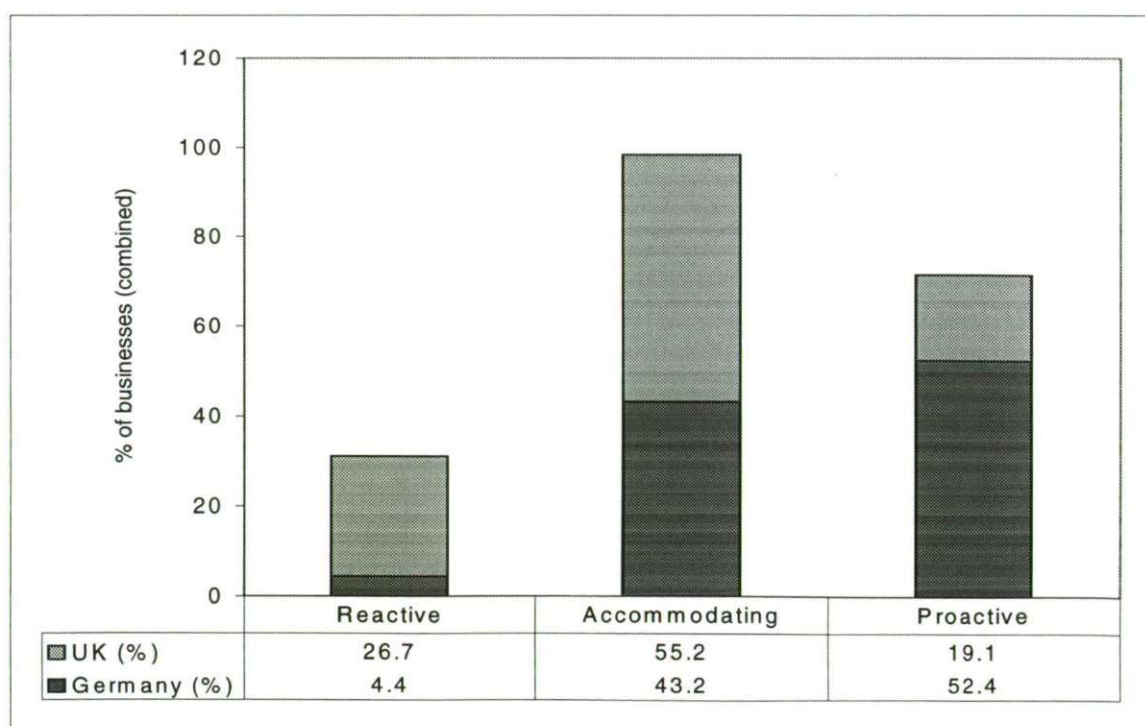


Figure 6.2 Company Classifications $WMHA_I$ Index

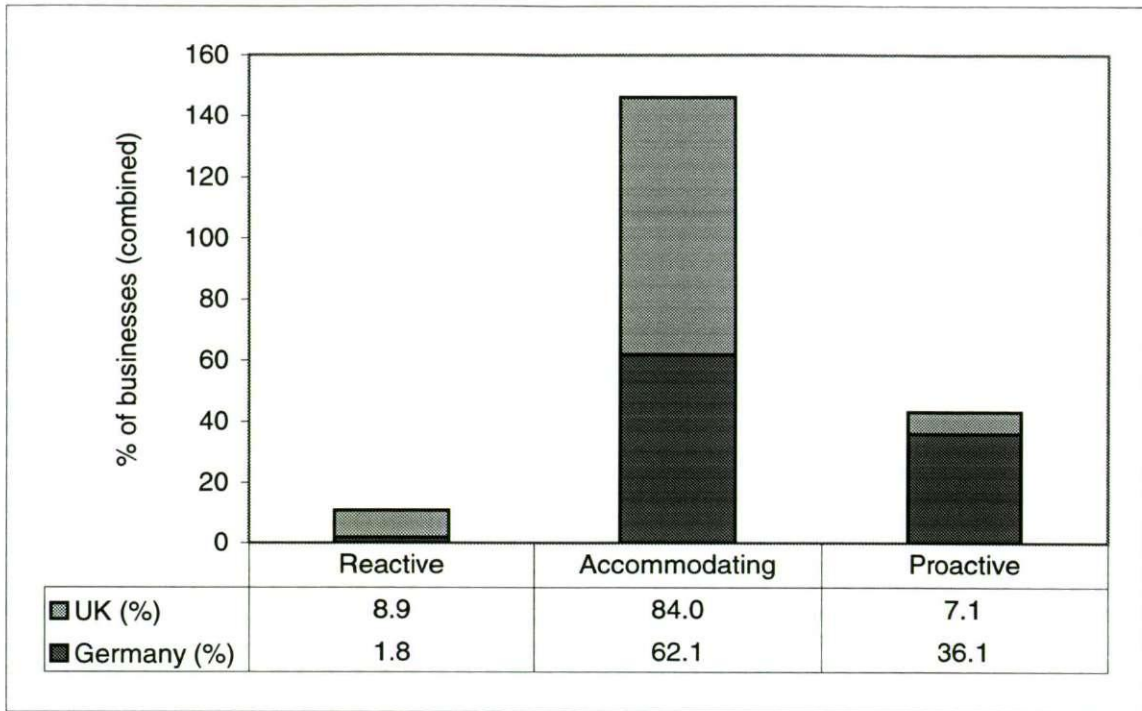
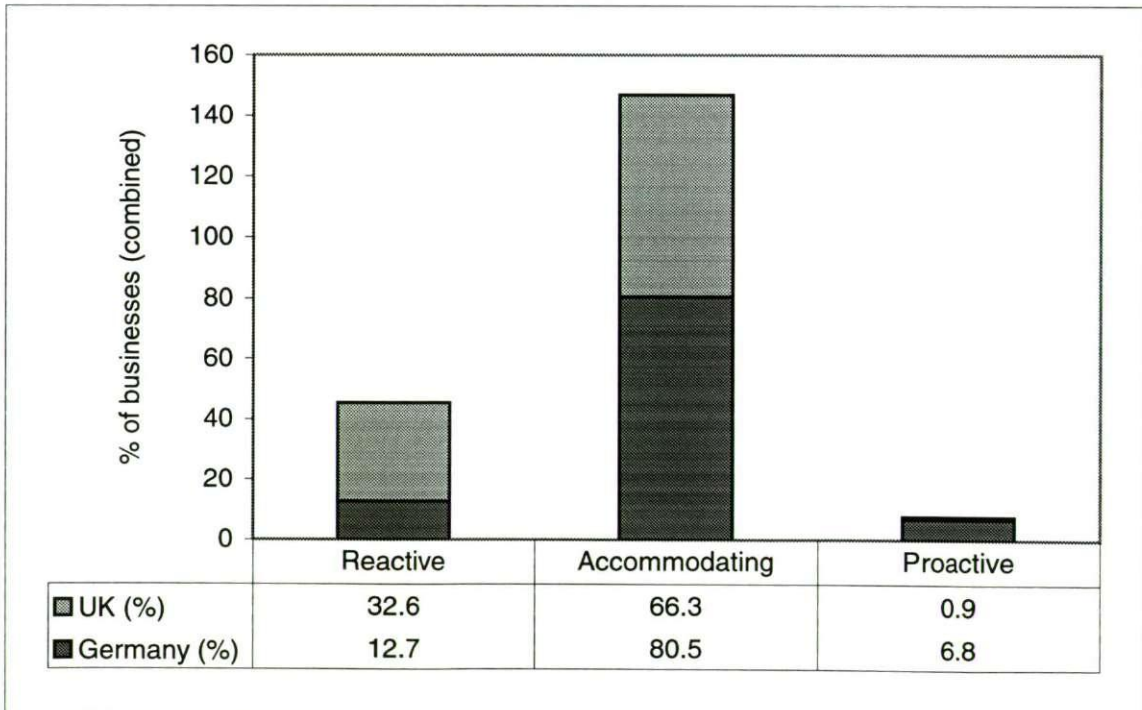


Figure 6.3 Company Classifications $WMHT$ Index



6.4.3 Waste Management Practices for Individual Packaging Materials

The final stage of the analysis was to assess business waste management practices for each packaging material covered by the Directive. The importance of this is underlined, first, by the problems highlighted in some UK reprocessing sectors (reported in Chapter five) and, second, by virtue of the fact that the production and waste management of each packaging material produces different environmental impacts. For example, plastics are considered to be highly environmentally damaging because of their petro-chemical composition and contribution to greenhouse gas emissions during manufacture and incineration. As composite materials, they are also difficult to recycle (Bailey, 1999b; MRW, 1999d). Conversely, some commentators are particularly concerned about the extraction of ore for aluminium production, whilst others highlight the ecological incongruities of policies which promote paper recycling before EfW incineration (Hanley and Slark, 1994; Collins, 1996)¹⁰. Any differences in the management of waste materials should therefore be recognised and evaluated. Although quantifying the precise environmental impact of each material accurately would require extensive and specialist LCA techniques, it was possible to develop a general picture of the Directive's impact for each packaging material using the waste hierarchy.

Considering the results of the previous analyses, it was expected that more German businesses would be engaged in preventative or closed-loop waste management for all packaging materials. Although this general trend was confirmed (Table 6.9), the main exceptions were the steel and wood sectors, where there was little difference between the re-use and purchase of recycled packaging in the two countries. This further reinforces the points, first from Chapter five, that UK steel and aluminium reprocessors have greater capacity for packaging than is required under the Directive and, second, that wood re-use and recycling figures have been distorted by the inclusion of wooden pallets within the UK Regulations.

Figures 6.4-6.7 illustrate the main waste hierarchy options employed for each packaging material by British and German producers, and re-emphasise the greater commitment of German respondents to preventative waste management. Against this

¹⁰ Hanley and Slark (1994) argue that EfW is preferable to recycling for paper, first, because paper can only be recycled a few times, second, because transporting waste paper involves excessive costs and environmental impact, and, finally, because recycling detracts from sustainable forest management.

general background, both industry groups appear to be more heavily engaged in preventative or closed-loop management for paper, plastics and wood than glass, steel and aluminium. The well-developed recycling networks for paper undoubtedly help explain the high levels of recycling in both countries, a factor that also accounts for the popularity of wood re-use in the UK.

Table 6.9 Waste Management Activity by Packaging Material

		<i>% Companies engaged in WMH activity</i>			
		<i>Reduction</i>	<i>Re-use</i>	<i>Collection</i>	<i>Purchasing Recycled Packaging</i>
Paper	UK	14.6	41.8	59.6	60.6
	Germany	55.8**	57.1**	89.1**	83.4**
Glass	UK	0.7	3.9	6.1	3.6
	Germany	11.7**	20.2**	35.1**	21.1**
Steel	UK	1.9	16.6	16.4	11.4
	Germany	6.6**	18.7	50.0**	10.6
Aluminium	UK	0.7	3.5	7.2	4.7
	Germany	8.6**	8.1**	20.8**	6.5
Plastics	UK	9.3	35.6	39.7	27.3
	Germany	40.1**	51.0**	77.2**	48.2**
Wood	UK	5.1	36.4	27.1	22.1
	Germany	14.2**	30.3	48.5**	18.1

** differences between UK and German waste management rates over 99% significant.

It should nonetheless be noted that there has also been a more widespread shift towards the prevention of paper waste amongst German businesses. This is also true for plastics, where over 40% of German businesses are engaged in packaging reduction, compared with fewer than 10% of British firms. Whilst the link between this and recycling charges is explored in Chapter seven, it initially appears that the punitive charges for plastics set by the DSD has persuaded German industry that minimisation is a more economic option than reclamation and recycling.

Figure 6.4 Collection of Packaging Materials

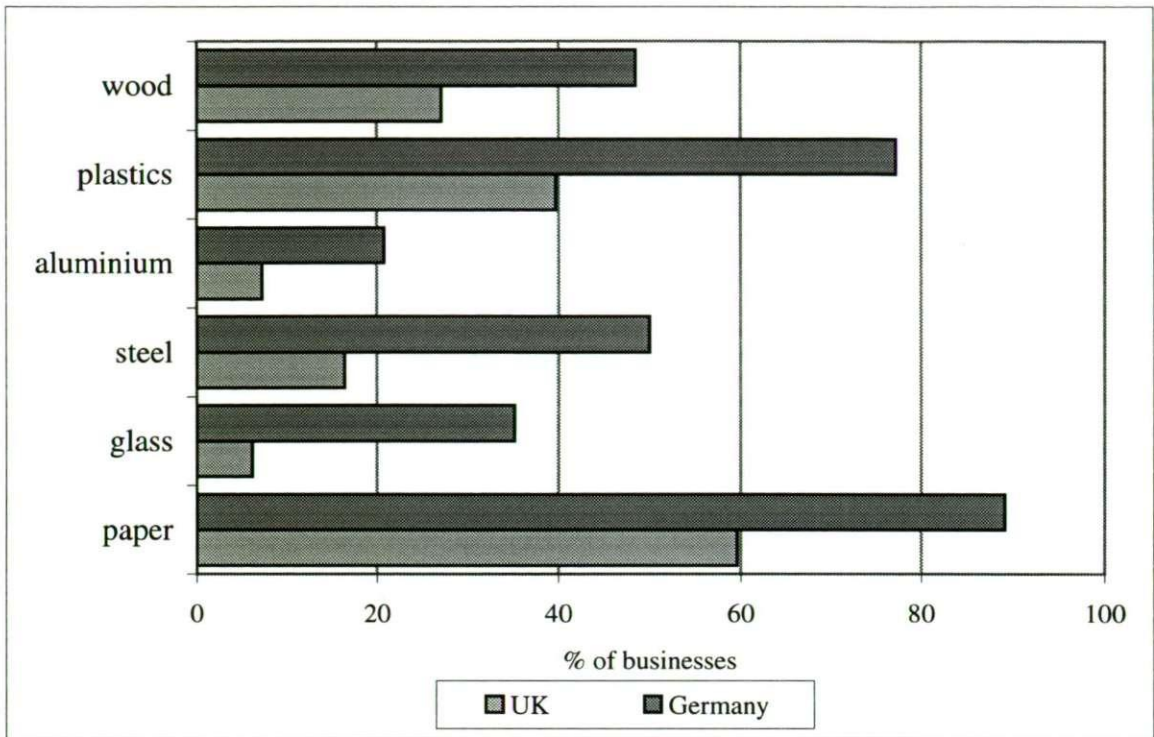


Figure 6.5 Reduction of Packaging Materials

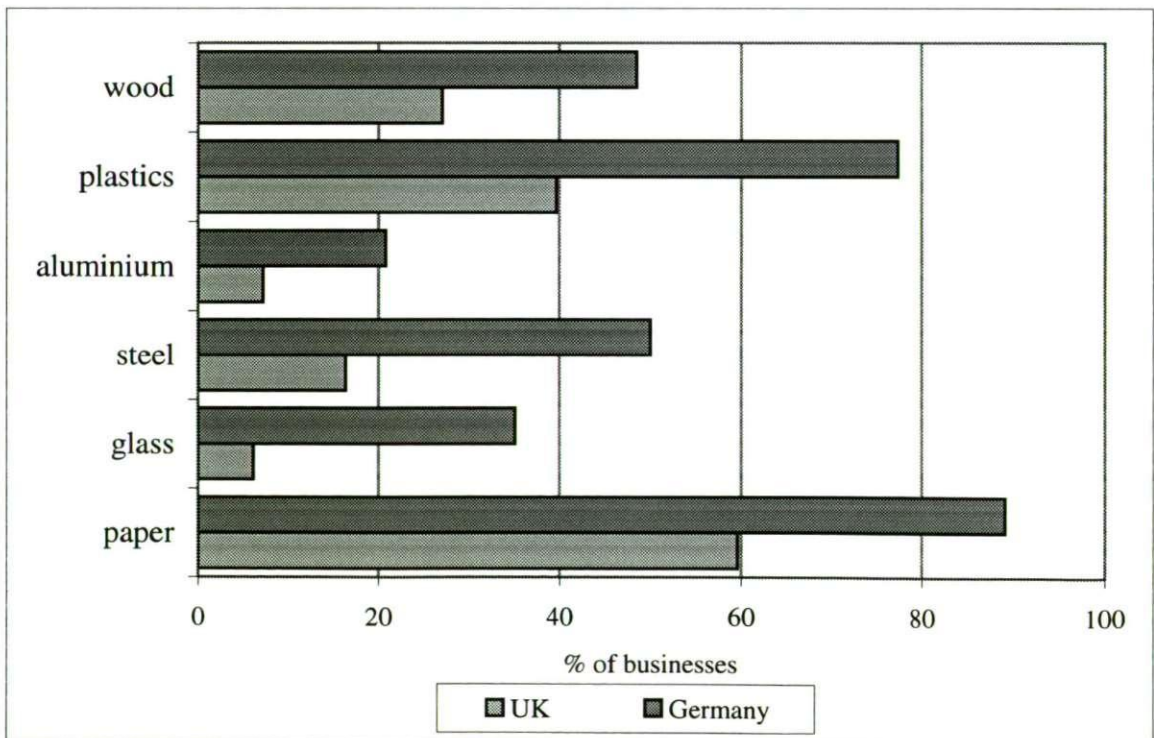


Figure 6.6 Re-use of Packaging Materials

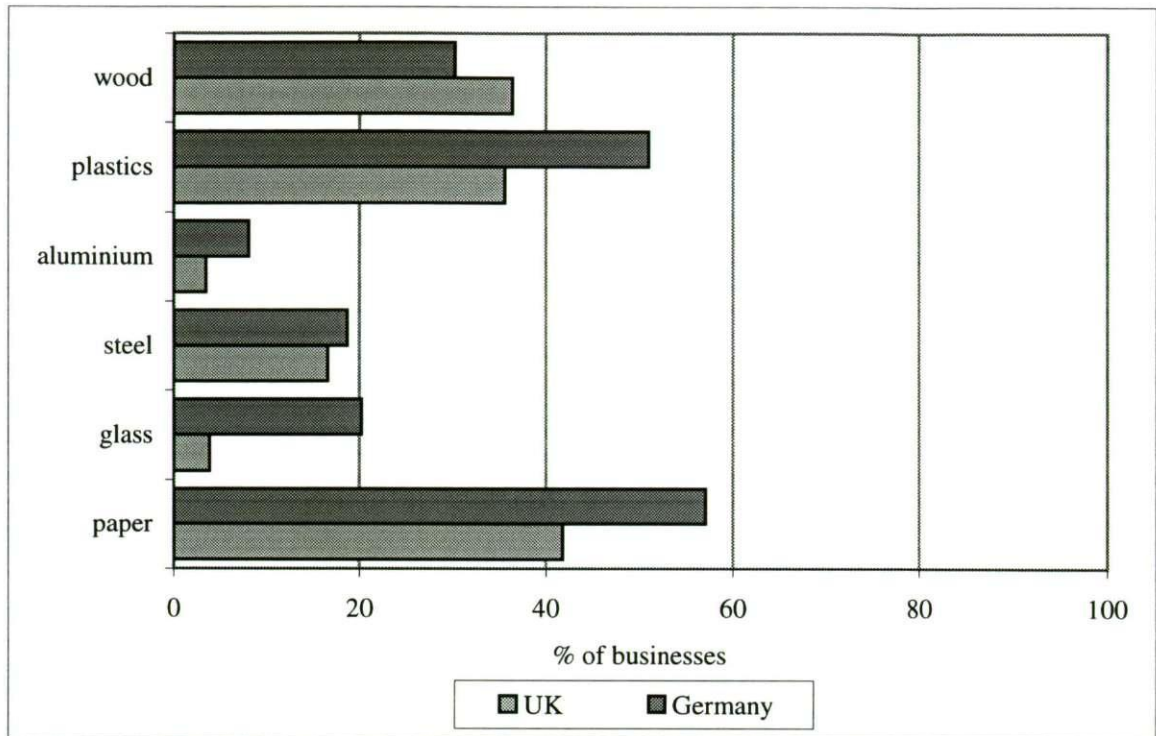
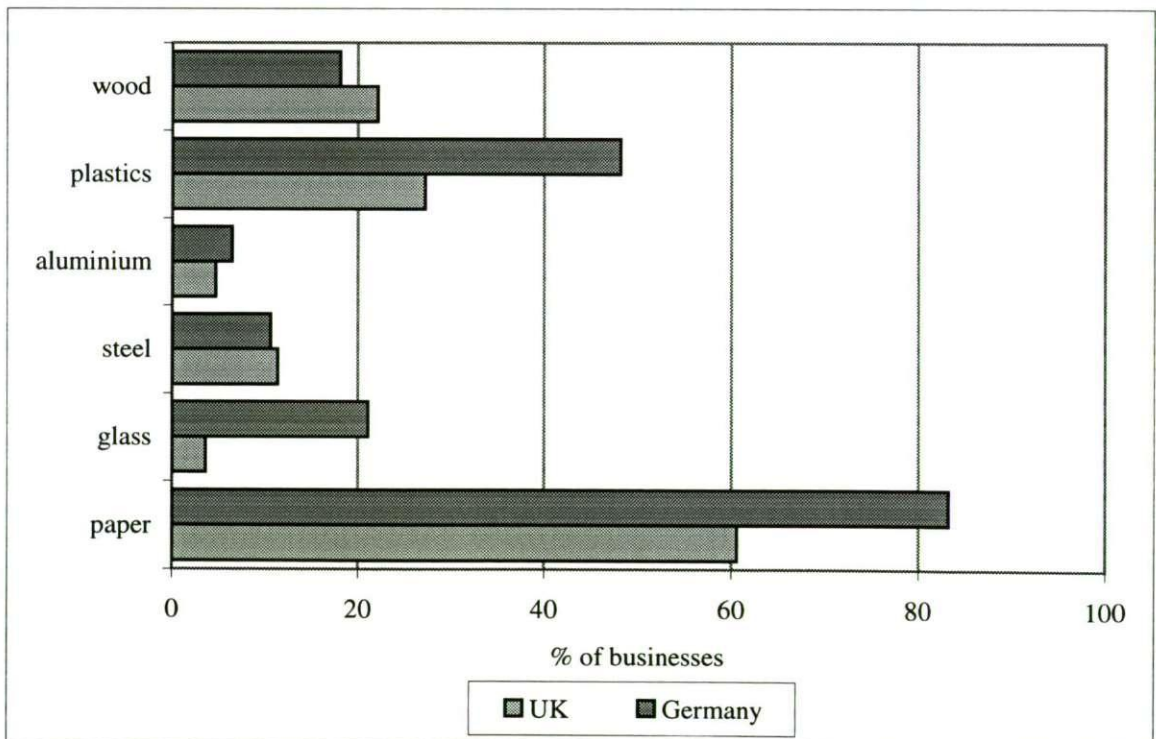


Figure 6.7 Purchase of Recycled Packaging Materials



From the limited data available, it is unsafe to make more than tentative claims about the prevailing waste management trends for each packaging material. This is particularly true considering the volatility of many materials markets (see Chapter five) (Eminton, 2000). Nonetheless, there appears to be a move away from end-of-pipe solutions for paper and plastics in Germany, whilst recycling and re-use seem to be the preferred strategies for steel, aluminium and glass. As yet, there is little evidence of UK businesses embracing preventative waste management to the same extent as their German counterparts, though materials re-use seems to have become more popular. Whilst the causes of such variances remain a matter of conjecture, four possible explanations stand out. The first is that business behaviour has been influenced by the higher recovery and recycling targets set by the Packaging Ordinance. Related to this is the fact that the Ordinance has been operating for a longer period of time. It is therefore likely that the influence of the legislation over industry behaviour has increased over this period (Bailey, 1999a). The third possibility is that a fundamental gulf exists between British and German industry in terms of social responsibility culture (Woolcock *et al.*, 1991; Knabe, 1995; Egan, 1997). Although this factor is outside the scope of this thesis and would be problematic to evaluate objectively, its influence should not be discounted. However, the explanation with the most profound implications for EU environmental policy and its desire to promote price-based regulation, is that the variance is a product of the environmental-charge systems operating in each country. The strength of this association is examined further in Chapter seven.

6.4.4 Qualitative Feedback

In addition to assessing the waste management actions adopted by the two respondent groups, the survey also sought to explore their general perceptions of packaging waste legislation. Four main themes were explored. The first aimed to establish whether managers believed that packaging regulation would change company waste management practices. The second asked whether businesses would achieve or exceed the minimum standards set by the Directive. The third asked whether the Directive would encourage reduction and re-use in addition to recycling. The final theme sought to determine whether businesses considered the Directive's recovery and recycling targets to be realistic. In each case, respondents were asked to rank their agreement or otherwise with given proposition statements using five-point Likert scales (see Chapter

four). Whilst such attitude data cannot be treated as 'hard data' in the same way as specific and measurable waste management actions, they help to clarify business' overall opinions on the efficacy of the Directive and its implementing methodology.

The results, shown in Tables 6.10 and 6.11, indicate that whilst there were again marked differences of opinion between German and British producers in some areas, in others there were noticeable similarities. For example, both groups agreed that the costs of compliance were encouraging changes in corporate practices and that the legislation would promote reduction and re-use as well as recycling. However, German businesses were significantly more positive about the environmental merits of the Directive. Greater contrasts again emerged in the degree of ambition expressed in German and UK waste management plans, where 77% of British companies foresaw themselves achieving the minimum statutory targets but 70.7% of German firms felt confident they could surpass them.

However, the data also revealed trends that require further explanation. Despite the overall optimism of German respondents, there was a general consensus that the Directive's recycling targets are unrealistic. At the same time, neither group felt their business had been unfairly discriminated against. This suggests that whilst both German and UK companies have reservations about mandatory recycling targets, neither group felt they had been unfairly penalised in comparison with other industry sectors. The fact that German businesses can express dissatisfaction but, at the same time be bullish about their recycling targets implies that they have become accustomed to operating within the strictures of the Ordinance. Thus, corroborating the typologies provided by the aggregated waste management indices, German companies are moving beyond compliance with environmental laws towards voluntary initiatives (also Bhargava and Welford, 1996). By contrast, British companies are still adjusting to their responsibilities and remain pessimistic about the future. Thus, whilst the behaviour of the German respondent group still falls short of Bhargava and Welford's attributes of environmental legislation 'transcenders' in that there is little evidence of them 'proactively engag[ing] in setting the agenda' (p. 21), their attempts to pre-empt possible future regulation should be seen as a significant, and apparently widespread, shift towards sustainable business thinking.

Table 6.10 Producer Opinions on the Directive, Mann Whitney analysis

<i>Proposition statement</i>		<i>N</i>	<i>Mean Rank</i>	<i>Sum of ranks</i>	<i>Mann-Whitney U</i>	<i>2-tailed sig.</i>
The cost of the Regulations (Ordinance) will encourage the Company to change its policies on the use of packaging	UK	426	289.09	123151.5	32200.5	0.000
	Germany	197	361.55	71224.5		
	Total	623				
The Company will only be able to comply with the minimum standards set by the Regulations (Ordinance)	UK	425	339.55	144309.0	38441.0	0.000
	Germany	217	286.15	62094.0		
	Total	642				
The Company aims to exceed the targets set by the Regulations (Ordinance)	UK	412	257.69	106168.5	21090.5	0.000
	Germany	215	421.90	90709.5		
	Total	627				
The Regulations (Ordinance) will encourage the Company to reduce and re-use more of its packaging than it would otherwise have done	UK	423	295.26	124893.0	35217.0	0.001
	Germany	198	344.64	68238.0		
	Total	621				
The recycling targets are too high for industry to achieve	UK	429	318.42	136602.0	39717.0	0.247
	Germany	196	301.14	59023.0		
	Total	625				
The Company has been unfairly discriminated against in the legislation	UK	430	303.67	130576.5	37911.5	0.033
	Germany	196	335.07	65674.5		
	Total	626				

Table 6.11 Producer Opinions on the Directive, Descriptive analysis

<i>Proposition statement</i>		<i>N</i>	<i>Mean</i>	<i>SE Mean</i>	<i>Standard deviation</i>	<i>Median</i>	<i>Mode</i>	<i>% of businesses agreeing with statement^a</i>
The cost of the Regulations (Ordinance) will encourage the Company to change its policies on the use of packaging	UK	426	0.20	0.06	1.15	1.0	1.0	53.0
	Germany	197	0.69	0.06	0.90	1.0	1.0	71.1
The Company will only be able to comply with the minimum standards set by the Regulations (Ordinance)	UK	425	0.32	0.06	1.13	1.0	1.0	53.7
	Germany	217	-0.02	0.08	1.16	0.0	1.0	40.5
The Company aims to exceed the targets set by the Regulations (Ordinance)	UK	412	-0.17	0.05	1.02	0.0	0.0	23.0
	Germany	215	0.78	0.06	0.88	1.0	1.0	70.7
The Regulations (Ordinance) will encourage the Company to reduce and re-use more of its packaging than it would otherwise have done	UK	423	0.22	0.06	1.14	1.0	1.0	51.5
	Germany	198	0.58	0.06	0.91	1.0	1.0	61.1
The recycling targets are too high for industry to achieve	UK	429	0.29	0.05	1.03	0.0	0.0	42.9
	Germany	196	0.19	0.07	0.92	0.0	0.0	37.3
The Company has been unfairly discriminated against in the legislation	UK	430	-0.06	0.05	1.02	0.0	0.0	24.2
	Germany	196	0.08	0.07	0.92	0.0	0.0	27.1

a % of all respondents with a score of 1 or 2 for the proposition statement

Whilst one might speculate whether a similar transition might occur in Britain once the UK Regulations have been operating for a longer period, such a move is logically dependent on the pattern of incentives created by the PRN system. The uncertainties and conflicts within the scheme, reported in Chapter five, and its uneasy coupling of economic and environmental objectives seem to have done little to make industry receptive to the goals of environmental sustainability. It therefore seems that despite the policy Europeanisation produced by the Directive, domestic political-economic agendas and implementation styles remain critical determinants of policy outcome. Thus, while Germany's determination to be at the vanguard of environmental policy is profoundly shaping the behaviour of its industries, the slow progress in Britain reflects the pragmatic and sometimes defensive style of UK environmental policy (Haigh and Lanigan, 1995). On the current evidence, there is some way to go before this approach convinces British industry that environmental stewardship should be a central objective of its business strategies.

6.5 Conclusions

A number of key facts about corporate waste management in Britain and Germany have been uncovered during the producer survey. First, substantially more German respondents are involved in preventative or closed-loop systems of waste management. Importantly, this process seems to have extended to the SME sector despite, or perhaps because of, the costs associated with the Dual System. Undeniably the key point of the analysis, however, was that German companies are moving towards preventative rather than 'end-of-pipe' methods of waste management. Whilst rigid interpretations of the waste hierarchy are heavily criticised for over-simplifying complex environmental and economic issues, there is almost universal agreement that source reduction is the only truly preventative form of resource management (Allaway, 1992; Lober, 1996; Barrett and Lawlor, 1997). The move by German businesses towards preventative management is particularly important considering the fact that it is only promoted in the Ordinance via the medium of price-based regulation.

However, similar distinctions can be made for all aspects of the waste hierarchy. It can therefore be argued that a far-reaching shift towards sustainable waste management is occurring in Germany but that a similar trend has yet to materialise in Britain. Even

where this distinction is less acute, for example, in relation to packaging re-use, it remains highly significant. A similar situation prevails for the management of more environmentally-damaging packaging materials, particularly plastics. Although Chapter five noted that both PWMSs impose their highest levies on plastics packaging, the difference in charge rates would appear to be a critical differentiating factor. Whilst it would be optimistic to claim that Germany is no longer a waste profligate society - Hagengut (1997), for example, shows Germany to be one of the EU's highest per capita packaging consumers - DSD's claim to have moved 'from a waste mountain to a waste shortage' does appear to have some credence (DSD, 1998: 5). The only materials which are apparently managed in parity are those where Britain already possesses a well-developed recovery and recycling infrastructure. In summary, British industry's stance seems to be one of reluctant compliance, whereas the Ordinance appears to have succeeded in integrating environmental principles into the waste management activities of obligated firms.

The question must therefore be why such divergent environmental outcomes have emerged from a process ostensibly designed to harmonise EU packaging laws. Although the EU policy process is clearly not designed to enforce total policy harmonisation (see Chapter three, also Bailey, 1999a), such disparities require serious examination if policy approximation and sustainable development are serious EU ambitions (Krämer, 1996; Demmke, 1997). Moreover, uncovering the reasons behind this apparently two-tier system, be they political, economic or simply temporal, should assist in the development of future environmental policies. Chapter seven now considers these issues by examining the influence of packaging waste charges on corporate behaviour.

Chapter Seven

The Role of Economic Instruments in Environmental Policy Implementation

7.1 Introduction

The first element of the survey highlighted marked contrasts between the waste management practices of British and German packaging producers. Whilst this was not surprising considering the distinct policy approaches adopted by the two governments (see Chapter five), this chapter explores whether differences in corporate environmental behaviour can be explained by the influence of environmental charges. The discussion is organised into the following sections. The first discusses the theoretical foundations of the economic approach to environmental management. The aim is not to repeat the discussion on the selection and application of environmental policy instruments from Chapter two, but rather to emphasise the key arguments underpinning the incentive effect of price-based regulation. The second section highlights the waste management costs incurred by businesses and their effect on corporate actions using further data from the producer survey. This is followed by an examination of producer opinions towards possible alternatives to the use of producer-related economic instruments (DTI/DoE, 1991; 1992; DETR, 1998a; 1999a; 1999b). On the basis of this evidence, a model of industry response to legislative and price-based regulation is developed and discussed.

7.2 Theoretical Background

The main difficulty of unregulated market systems in relation to environmental resources is that the full social costs of their utilisation are rarely reflected in market prices (Pearce *et al.*, 1989). This creates a tendency towards over-exploitation and, it is argued, economic inefficiencies (Devlin and Grafton; 1998). The economic viewpoint maintains that correcting such distortions requires the introduction of compensatory market mechanisms which re-internalise externality costs (Turner, 1993; Burningham and Davies, 1995). Brisson (1993) and also Pearce and Turner (1992; 1993) make this point specifically in relation to the proliferation of packaging waste.

The principal aim of environmental taxes, therefore, is to provide producers or consumers with a financial incentive to reduce their environmental damage. The ambition is that pollutants should be stabilised at 'socially-optimum' levels - the point where the cost of further abatement exceeds the benefit gained by society (measured in terms of total social utility) (Brown *et al.*, 1995; van den Bergh, 1996). Two other major benefits are also said to accrue from the economic approach. The first is the so-called 'double dividend'; the stimulation of environmental industries and resource-efficient production techniques. The second is the hypothecation of environmental tax revenues, either for environmental expenditures or to reduce taxation on economic 'goods', such as employment (Gee and von Weizsäcker, 1994, Bohm, 1997; Spackman, 1997).

Most supporters of environmental valuation accept that economic instruments supplement rather than replace legislative standards (Acutt and Dodgson, 1997; O'Doherty and Garrod, 1999). However, the critical distinction between the two techniques is that legislative standards only impose threshold constraints on pollution (the standards contained in the legislation), while economic instruments can be used to monetise each unit of pollution, thereby creating a constant pressure for improvement.

Polluters pay not just the cost of reducing pollution to the acceptable level (which is what they would be required to do under a regulatory system), but also an additional sum on top of this. The additional sum arises because the tax is paid on all the damaging activity, not just the proportion above the target level. This sum can be seen partly as a payment for the residual damage caused by the pollution at the target level, and partly as a 'rent' on the use of the environment. The important point to note is that the payment of the extra amount effectively *removes* rights that the polluter enjoys under a regulatory system (Jacobs (1991: 160, original emphasis).

Furthermore, where economic instruments utilise market forces as part of their abatement strategy, pollution costs can also be made to respond dynamically to changing economic and environmental circumstances (Goddard, 1995). The notion of an incentive effect has become a central orthodoxy of many environmental-valuation techniques. Pearce *et al.* (1993: 96), for example, notes that:

Since price is instrumental in changing behaviour it follows that taxation policy will also be an important influence on behaviour which affects the environment (emphasis added).

Similarly, Baumol and Oates (1979: 231) remark that:

In each of these cases [the introduction of environmental charges or subsidies] the basic notion is the same: by offering virtue its just (financial) reward, we change the rules of the game to induce industry (and individual consumers) to alter their behaviour to promote an environmental objective.

Hahn (1989: 95), stresses both the environmental and economic benefits of price-based regulation:

Both [marketable permits and emission charges] represent ways to induce businesses to search for lower cost methods of achieving environmental standards. They stand in stark contrast to the predominant "command-and-control" approach in which a regulator specifies the technology a firm must use to comply with regulations.

The behaviour-changing potential of economic instruments is not undisputed, however. Jacobs (1991) argues that whereas legislation compels firms or households to observe pre-defined environmental standards, incentives merely encourage them to do so. Ultimately individuals may choose to pay the charge and continue polluting as before (see also Pezzey, 1993). The point at which environmental charges effectively control a particular pollutant therefore depends on the price elasticity of the product or process. Consequently, a number of charge iterations may be necessary before the desired abatement is achieved. Pearce and Barbier (2000) suggest that environmental valuation techniques can overcome these obstacles but Dickens (1996) and More *et al.* (1996) question the competency of economics outside its core thematic areas.

Jones (1999) further suggests that as businesses usually make investment and technology decisions on the basis of total factor costs rather than constituent components thereof, the relationship between environmental taxes and business behaviour should not be over-exaggerated (also Cairncross, 1991). Hahn (1989: 95) concedes that the theoretical structure of environmental economics 'often emphasises elegance at the expense of realism' and Jacobs (1991: 152) that many models: 'fail to

represent the complexities of the real world, in which “institutional” factors crucially affect corporate and consumer decision-making.’

The empirical evidence of the incentive effect is also somewhat uncertain. Opschoor and Vos (1989) and Hahn (1989) argue that environmental charges have failed to create significant pollution-reducing incentives but have the ‘compelling virtue’ (Tietenberg, 1990: 32) of achieving targets in a cost-effective manner. Similarly, Barde (1997) claims that most existing environmental taxes are still too low to alter polluter behaviour. He asserts, for example, that carbon taxes must be set significantly above \$50 per tonne if CO₂ emissions are to be stabilised at their 1990 levels by 2050 (Sweden, a world leader in environmental taxes, currently charges polluters \$41 per tonne for carbon emissions). Thus, there is an obvious sufficiency requirement for price-based environmental regulation. In defence of the economic approach, Hahn (1989), Huppel *et al.* (1992) and Goddard (1995) note that charges and marketable permits are rarely introduced in their textbook form but are instead often inappropriately grafted onto regulatory systems where standards play a dominant role.

The incentive effect of price-based regulation is therefore an important, if controversial, element of the economic approach to environmental policy. The literature has both defended the need to monetise environmental resources and recognised practical weaknesses of this methodology. The remainder of the chapter assesses the merits of the debate by examining the relationship between packaging charges and producer behaviour in Britain and Germany.

7.3 Costs Incurred by Packaging Producers

In enumerating the costs incurred by industry as a result of packaging regulation, it is recognised that environmental charges are only one component of total corporate liability. This is well recognised in the literature, where indirect costs are classified as a form of transaction cost (see Tietenberg, 1990; Jacobs, 1991). The phenomenon of indirect costs was also identified in the DoE’s criteria for evaluating policy instruments (DoE, 1993, see Chapter two). However, despite the fact that informal compliance costs are difficult to calculate accurately, some authors suggest they are a major element

of the total expenditure associated with environmental legislation (Baumol and Oates, 1988; Beder, 1996).

Two techniques were used in the survey to establish the nature and extent of producers' indirect compliance costs. The first was to provide a list of suggested cost areas, both formal and indirect, whilst also inviting respondents to add further categories. The second was to quantify the labour resources devoted to compliance activities, such as data gathering and contract management (Walley and Whitehead, 1996). The merits of asking respondents to quantify the value of their indirect costs were also considered. However, it was decided that such data would contain many 'guesstimates' and, therefore, would be too subjective for evaluation purposes. The development of techniques that accurately categorise and quantify the indirect compliance costs sustained by businesses in relation to environmental policy would nonetheless be of great benefit to future research in this area.

Table 7.1 Compliance Costs Incurred by Producers

<i>Cost Category</i>	<i>% of Companies incurring cost</i>		
	<i>Germany</i>	<i>UK</i>	<i>Chi-Square (χ^2) and sig.</i>
Enforcement Agency registration fee	16.24	59.17	100.739 0.000
Compliance scheme membership fee	37.06	72.71	72.915 0.000
Per packaging unit recycling charge	60.41	51.15	4.684 0.030
Deposit-refund charges on packaging	9.64	1.61	22.265 0.000
Green Dot charges on sales packaging	59.90	2.98	267.832 0.000
Fees to external consultancies	10.66	7.34	1.950 0.163
Higher waste collection costs	61.42	30.05	55.746 0.000
PRNs or related compliance certificates	9.14	38.30	55.802 0.000

Table 7.1 summarises the compliance costs incurred by British and German respondents. As few respondents mentioned cost categories not already suggested in the questionnaire, it was assumed that the majority of possible compliance expenses had been correctly identified. This data in fact revealed few trends not anticipated from previous analysis of the British and German regulations (Chapter five). As expected, the majority of German businesses were incurring Green Dot or other per unit packaging charges (related to reprocessing fees as well as collection costs for transport and secondary packaging) and British firms were mainly paying Environment Agency, compliance scheme or PRN fees. Similarly, the low number of businesses engaged in deposit-refund systems reflects the deferral of the Ordinance's take-back provisions and their exclusion from the Regulations (Michaelis, 1995). The most important finding, therefore, was that many companies are incurring higher waste collection costs. As there was again a significant difference between the two groups - 61.4% of German firms and 30.1% of British respondents reported higher collection costs - this may be a factor causing the different patterns of waste management in Britain and Germany. Resource constraints prevented a further survey specifically to examine this point but it should be relatively simple to quantify in future research.

Table 7.2 Staff Employed to Manage Compliance with Packaging Regulations

<i>Staff employed</i>		<i>% of businesses</i>					<i>Mann-Whitney & 2-tailed sig.</i>
		<i>0</i>	<i>1</i>	<i>>1-2</i>	<i>>2-5</i>	<i>>5</i>	
Total	Germany	9.0	1.6	64.0	19.5	5.9	
	UK	46.3	13.2	36.0	3.7	0.8	15185.0 0.000
Additional	Germany	90.5	6.0	3.5	0.0	0.0	
	UK	94.0	4.0	1.8	0.2	0.0	41359.5 0.108

The impact of packaging regulation on employment costs (Table 7.2) also revealed significant differences between the two respondent groups. Whilst 65.6% of German companies and 49.2% of British firms had 1-2 members of staff devoted to managing compliance activities, 46.3% of UK respondents and 9% of German businesses had no

staff dedicated to this function. However, the vast majority of companies in both groups claimed to have employed no additional personnel (90.5% in Germany, 94% in Britain). This indicates that managing compliance with packaging legislation has not proved sufficiently burdensome that either group has needed to engage additional staffing resources. This is despite vociferous complaints about the excessive complexity of the UK Regulations (ACP, 1998). The suggestion was further reinforced by the small number of firms engaging external consultants to advise on the intricacies of compliance, where 75.9% of German and 78% of British respondents were managing this task internally ($\chi^2 = 0.648$, significance = 0.421). The probable explanation for this is that national environment ministries, enforcement agencies, the DSD and the UK compliance schemes have produced a wealth of material relating the compliance options open to obligated companies (DETR, 1997a; Environment Agency, 1997; VALPAK's quarterly *Vantage* magazine; DSD, 1999c; <http://gruener-punkt.htm>). It also suggests that ongoing employment costs have not been a significant determinant of corporate responses to packaging waste regulation.

7.4 The Relationship between Environmental Charges and Producer Behaviour

Whilst it was always unlikely that a postal survey could identify the full range of packaging waste costs incurred by obligated businesses, the main weakness in the data was the failure to quantify waste collection costs. Recognising this limitation, the remainder of the section examines the effect of environmental charges on business waste management practices. Again the evaluation criteria are based upon the key objectives of the Packaging Directive and the waste management hierarchy.

The strength of the relationship between charges and waste management actions was tested using the *Cost-burden vs. Action (CbvA)* indices developed in Chapter four. To re-cap briefly, these indices factor out company size (in terms of turnover and number of employees) by dividing compliance costs by the midpoints of the categories used in questions E1 and E2 of the survey (Appendix 3). Thus, the *CbvA* indices measure the association between relative compliance-cost burden and each waste management action. *CbvA* indices were also calculated for the consolidated *WMHA*, *WMHA₁* and *WMHT* statistics (see Chapter six). Based on the assertions of the environmental economics literature, the initial hypothesis was that there would be a positive

correlation between packaging waste costs and at least some waste management variables. It was expected that these would be more pronounced for German producers because of the higher recycling fees in existence and their application to a greater proportion of the packaging used by respondent companies.

Table 7.3 Correlation of Waste Management Actions and Costs (*CbvA*), Germany

Waste Management Variable	N	Turnover		Employees	
		<i>Spearman correlation</i>	<i>1-tail sig.</i>	<i>Spearman correlation</i>	<i>1-tail sig.</i>
Packaging Collection %	91	-0.215	0.021*	-0.186	0.045*
Packaging Reduction %	104	-0.014	0.446	0.040	0.350
1997 Packaging Re-use %	93	-0.107	0.154	-0.154	0.079
2001 Packaging Re-use %	85	-0.014	0.448	-0.070	0.260
1997 Buy Recycled %	90	0.101	0.173	0.149	0.092
2001 Buy Recycled %	84	0.077	0.244	0.106	0.183
<i>WMHA</i> Index	118	-0.174	0.029*	-0.187	0.026*
<i>WMHA_J</i> Index	118	-0.101	0.139	-0.085	0.189
<i>WMHT</i> Index	117	-0.023	0.403	-0.051	0.302

However, the results (Tables 7.3 and 7.4) reveal no clear relationship between compliance costs and any of the key waste management variables for either country. In fact, the only significant correlations were negative, a key example being that between packaging collection and *CbvA turnover* for Germany (Table 7.3). Whilst this might be taken to mean that businesses with lower relative compliance costs are more actively engaged in waste management, the associations were too weak to lend real credibility to such a statement (the strongest was just -0.215 for *CbvA turnover* in Germany)¹. It was therefore decided that an additional test was necessary to establish if there was at least a significant relationship between packaging waste costs and *whether* (as opposed to the

¹ Where +/-1 signified perfect positive or negative correlation and any relationship less than +/-0.4 was considered too weak to be indicative of a strong association. The fact that correlations as low as +/-0.098 were *statistically* significant reflects the large data sets used (Shaw and Wheeler, 1994).

extent to which) businesses are engaged in each stage of the waste hierarchy. Again the results proved inconclusive. Figures 7.1 and 7.2 show that while German businesses with higher comparative compliance costs are more inclined to collect waste and those in Britain are more likely to engage in waste reduction, the general relationship between costs and producer actions still appears very weak.

Table 7.4 Correlation of Waste Management Actions and Costs (CbvA), Britain

Waste Management Variable	N	Turnover		Employees	
		<i>Spearman correlation</i>	<i>1-tail sig.</i>	<i>Spearman correlation</i>	<i>1-tail sig.</i>
Packaging Collection %	284	-0.052	0.192	-0.098	0.049*
Packaging Reduction %	334	0.061	0.133	0.043	0.214
1997 Packaging Re-use %	273	-0.005	0.466	-0.049	0.206
2001 Packaging Re-use %	259	0.017	0.391	-0.018	0.388
1997 Buy Recycled %	271	0.103	0.045*	0.013	0.416
2001 Buy Recycled %	240	0.081	0.106	-0.039	0.271
WMHA Index	338	0.106	0.026*	-0.004	0.467
WMHA ₁ Index	325	0.080	0.080	-0.037	0.249
WMHT Index	353	0.004	0.471	-0.033	0.270

* Significant at 95% confidence

The study therefore failed to establish a clear association between the imposition of packaging charges and active business engagement in waste management. This is despite the evidence from Chapter six that German respondents have significantly greater involvement in recycling than their British counterparts and the numerous studies which show packaging consumption to have fallen substantially in Germany since the Ordinance was introduced (*Gesellschaft für Verpackungsmarktforschung*, 1996; Eichstädt *et al.*, 1999). The tentative conclusion, therefore, is that the differences between the two respondent groups have not been produced by environmental charges but are the result of other regulatory factors.

Figure 7.1 Waste Management Actions and Costs, Germany

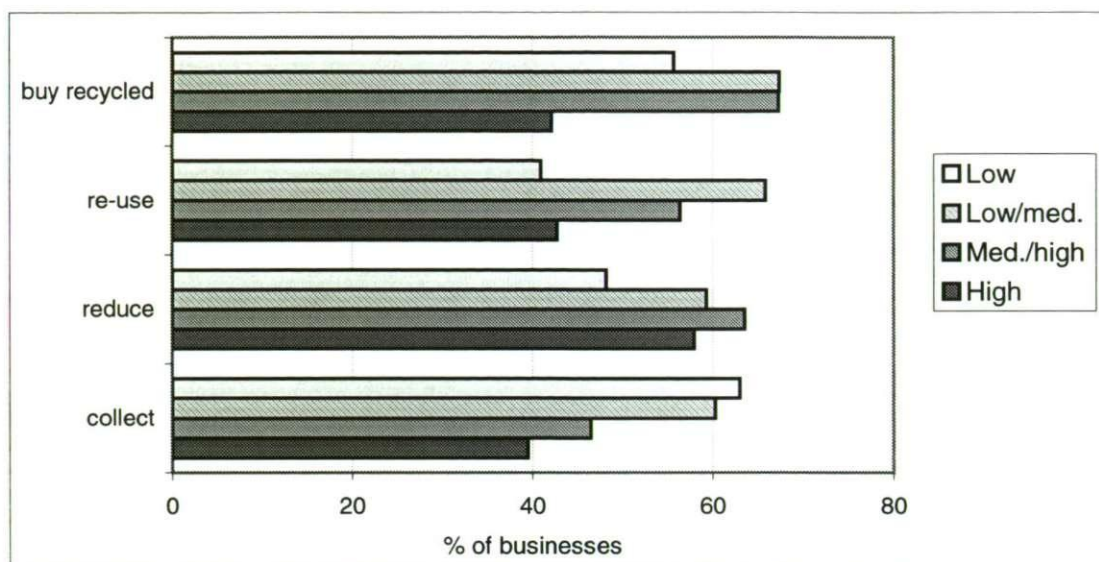
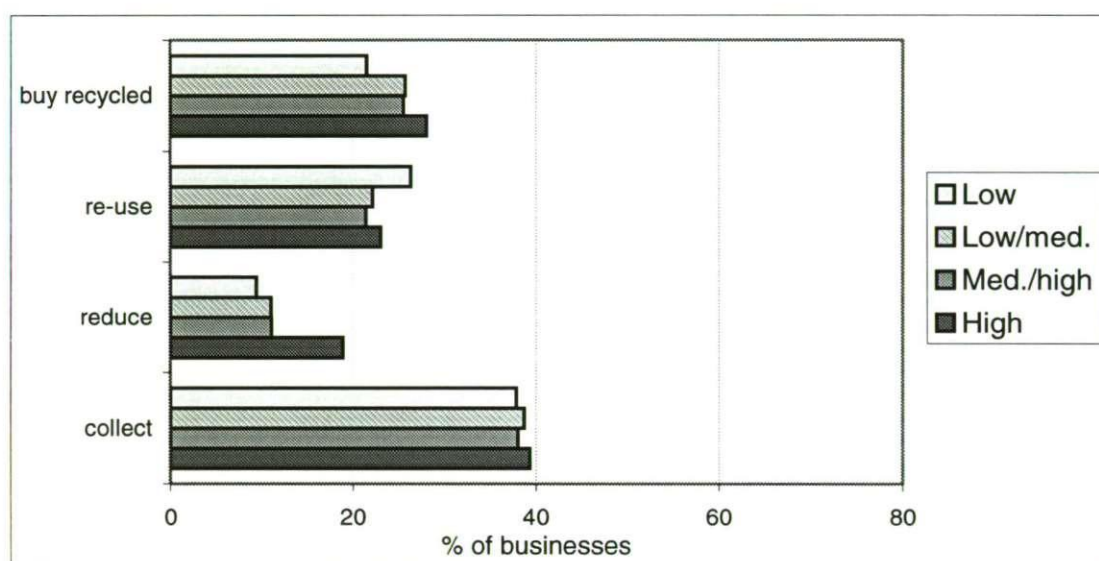


Figure 7.2 Waste Management Actions and Costs, Britain



7.5 Policy Alternatives

Although environmental charges form the mainstay of most European PWMSs, both the German and British governments have considered the use of other policy instruments to increase the economic and environmental efficacy of their recycling schemes. The 1998 review of the Regulations, in particular, clearly implied that additional mechanisms would be introduced if industry defaulted on EU recycling commitments (DETR, 1998a). While the debate has been less intense in Germany,

discussions have taken place on methods to improve the economic efficiency of the Dual System (Staudt, 1997; Flanderka, 1998).

In order to test business reactions to supplementary policy instruments, a list of suggested alternatives was compiled from government policy documents and reformulated into proposition sets (DETR, 1998a; 1999a; 1999b). Respondents were then asked to evaluate the proposed strategies using five-point Likert scales. The scales were again ranked from -2 (strongly disagree) to +2 (strongly agree). The proposition sets solicited opinions on:

- The use of voluntary agreements in place of legislation
- The abolition of formal packaging waste charges in favour of market-led initiatives to allocate resources
- The specification that packaging should contain a minimum percentage of recycled material
- The introduction of direct packaging waste charges for consumers
- Increased expenditure on consumer education (see also Bailey, 1999b; 2000)

In addition to these, respondents were asked their general opinions on the economic and environmental effectiveness of the regulatory mechanisms employed. Again proposition sets were provided to examine whether respondents thought:

- The current legislation and implementing mechanisms would achieve would achieve cost-effective solutions to the problem of packaging waste
- The regulatory regime would produce worthwhile environmental benefits
- Packaging waste charges were designed to cover the operational costs of recovery and recycling
- Packaging waste charges reflected the full environmental impact of packaging waste

Analysis of the proposition sets involved two stages. One-sample t-tests were first conducted to establish whether the mean responses were significantly above or below zero (the neutral opinion point). Mann-Whitney tests were then used to compare the opinions of the two respondent groups. The results are summarised in Tables 7.5 and 7.6. Table 7.5 shows that both respondent companies were generally indifferent to or

opposed the majority of alternative strategies suggested. First, UK businesses disagreed with the idea of replacing legislation with voluntary targets (D1), a tacit admission perhaps that recycling would not be taken seriously without formal regulation. However, German respondents were less opposed (though generally neutral) to voluntary targets, a possible reflection of the strain the Ordinance has placed on industry. By contrast, German firms refuted the idea that recycling targets could be achieved effectively without a formal pricing mechanism - an issue British respondents were undecided on - despite the high cost of the Dual System (D2). However, both groups were ambivalent towards the introduction of recycled-content quotas for packaging (D3). This result was somewhat surprising, particularly considering UK industry's strong opposition to this measure during the 1998 review of the Regulations (ACP, 1998).

Whilst British firms were largely neutral to the proposal (again submitted at the 1998 review) that consumers should be charged directly for packaging waste (D4), German respondents opposed the idea. The only policy alternative both groups favoured was increased expenditure on public education (D5). This feeling was particularly pronounced in the British group, reflecting industry frustration at the lack of onus on consumer participation in the UK Regulations. In terms of their overall appraisal, the UK group was adamant that the Regulations were not cost effective (D6). This was an issue German respondents were largely undecided on. Considering the higher cost of the Ordinance, this result might be considered unusual. Furthermore, German firms were clear that the Ordinance would bring worthwhile environmental benefits (D7) and both groups agreed that packaging charges reflected the operational costs of recovery and recycling (D8). However, their opinions differed on whether the PRN or Green Dot economic instruments considered the full environmental impact of packaging waste (D9).

This analysis therefore revealed two important facts about each group's perception of their current regulatory regime. Firstly, there seems to be little support for alternative modes of regulation. This is generally consistent with the fact that industry managers in both countries provided considerable input on the *modus operandi* for packaging regulation. Industry's acceptance of the overall regulatory framework (or, at least, fear

Table 7.5 Policy Alternative Proposition Sets

Measure	N	Descriptive Statistics			T-test a	Mann-Whitney U			
		Mean	Standard error	Standard Deviation	Sig. of variance from 0	Mean Rank	U statistic	2-tailed sig.	
D1 Industry voluntary targets would be more effective than legislation	UK	429	-0.31	0.057	1.18	0.000	303.73	38065.3	0.030
	Germany	198	-0.12	0.083	1.16	0.161	336.26		
D2 Recycling targets could be more effectively achieved without packaging waste charges	UK	419	-0.09	0.050	1.03	0.064	320.69	37003.0	0.017
	Germany	199	-0.33	0.068	0.96	0.000	285.94		
D3 Government should specify recycled content quotas for packaging	UK	416	-0.07	0.059	1.21	0.223	309.22	40051.0	0.642
	Germany	197	-0.12	0.087	1.21	0.161	302.30		
D4 Consumers should be directly taxed for packaging waste	UK	426	0.05	0.061	1.27	0.445	331.06	33415.0	0.000
	Germany	196	-0.40	0.093	1.30	0.000	268.98		
D5 More money should be spent on public education	UK	427	1.08	0.042	0.86	0.000	356.18	22980.5	0.000
	Germany	196	0.19	0.082	1.15	0.019	215.75		

a One sample t-tests could be used to compare variances from zero (neutral attitude), as the technique is reasonably robust to non-normal distribution. However, Mann-Whitney tests were used to compare the two groups because the data distribution for some variables was significantly skewed.

Table 7.6 Producer Assessment of Packaging Regulations

Measure		N	Descriptive Statistics				Mann-Whitney U			
			Mean	Standard error	Standard deviation	Sig. of variance from 0	Mean Rank	U statistic	2-tailed sig.	
D6	National regulations will achieve cost-effective solution to packaging waste problems	UK	427	-0.60	0.051	1.06	0.000	283.82	29811.0	0.000
	Germany	217	0.083	0.069	1.02	0.232	398.62			
D7	National regulations will produce worthwhile environmental benefits	UK	427	0.040	0.055	1.13	0.468	300.54	36954.0	0.000
		Germany	219	0.46	0.065	0.97	0.000	368.26		
D8	Packaging charges are based on operational costs of collection and reprocessing	UK	411	0.39	0.048	1.13	0.000	314.64	40428.0	0.281
		Germany	207	0.26	0.077	1.10	0.001	299.30		
D9	Packaging charges reflect full environmental impact of packaging production, use and disposal	UK	396	-0.11	0.050	1.00	0.024	283.11	33505.5	0.004
		Germany	197	0.13	0.075	1.05	0.078	324.92		

of more constrictive measures) was also exemplified in the German retailing sector's support for the Dual system during the 1993 crisis (see Chapter five). Furthermore, in line with Lévêque's (1995; 1996a) typology of business responses to environmental regulation, British industry seems latterly to have focussed on securing relative sectoral gains rather than disputing the need for regulation². Secondly, British businesses are more pre-occupied with the financial impact of the PRN system than German firms are with the ramifications of the Green Dot. This would suggest that British firms are still adjusting to the additional costs imposed by the PRN scheme - a transition undoubtedly made more painful by the recent high value of sterling - and that they are less bound into a co-operative '*Ordnungspolitik*' relationship with government (see Chapter three). However, the general opinion in both countries seems to be that there should be no major changes to existing legislation but that the emphasis should be on improving its enforcement and equity (ENDS, 1998e; 1998f).

The second stage of the producer survey therefore raises three fundamental questions. Firstly, if the differences between corporate waste management practices in Britain and Germany cannot be explained by the influence of environmental charges, what factors have caused this variation? Secondly, why have economic instruments failed to produce their desired incentive effect? Finally, what are the links between economic instruments and sustainable development if their incentive effect is less powerful than previously thought? The following section explores these questions.

7.6 Discussion of Survey Results

7.6.1 Factors Causing Variation in Waste Management Practices

A number of factors were highlighted in Chapter six as contributing to the different environmental outcomes engendered by the Ordinance and the Regulations. These included the fact that the Ordinance has been in force nine years compared with the three years the Regulations have been operating, the existence of higher recycling standards and strong government coercion in Germany, the degree of social-responsibility culture extant in each country, and the influence of environmental

² Examples include the provision of independent waste collection networks in supermarket car parks to compete with those managed by local authorities (ENDS, 1997d), moves by brewers to pay for bottle collection direct from pubs (ENDS, 1997e), and escalations in the use of transport packaging in order to increase the amount of waste which can be readily recovered by obligated businesses (ENDS, 1996).

charges. As price-based regulation has apparently prompted little positive response from industry, how have other factors affected the two policy outcomes?

Temporal factors

As the most obvious distinction between the Ordinance and the Regulations, it seems self evident that the time allowed for implementing mechanisms to become embedded will have a major impact on the environmental results achieved. However, whilst the UK Regulations are clearly still evolving rapidly, time may not be the critical factor. The Ordinance produced a four-fold increase in recycling in its first full year of operation according to official figures (DSD, 1998), though Staudt (1997) argues that recycling rates were already increasing in Germany prior to the Ordinance (see Chapter five). A second indication comes from the DETR's concerns that Britain will fall short of the Directive's targets, even allowing for the transition period provided for in the Regulations (DETR, 1999a). Therefore, whilst time factors are obviously important, they seem to be only a partial explanation.

The stringency of the legislative regime

Clear distinctions were also drawn in Chapter five between the legislative standards applied in Britain and Germany. Even following the 1998 revisions, the Ordinance contains targets higher than those set by the Directive and threats of further coercive measures if these are not achieved. Some DETR officials have privately suggested that the UK Regulations would induce a more active response from industry if recovery and recycling targets were raised to similar levels as those in Germany (O'Doherty and Bailey, 2000). However, VALPAK opposes this and argues that the government should first clamp down on free riders and extend recycling obligations to smaller firms (MRW, 2000a; 2000b).

It would therefore appear that the state's ability to direct legislative and implementing mechanisms plays an important part in the attainment of environmental objectives. As Jordan (1999: 69) (quoting Joseph Stalin) notes: 'To govern is not to write resolutions and distribute directives; to govern is to control the implementation of the directives.' Both Germany and Britain have been forced to reconsolidate their packaging systems, Germany to redress the DSD's financial crisis and the deficit in reprocessing

infrastructure, the UK to improve the efficacy of PRN hypothecation arrangements (see Chapter five). However, the centralised structure of the Dual System enabled the German government to adjust both its regulatory controls *and* Green Dot prices but the British government's (and industry's) preference for market-based pricing has severely curtailed the intervention options available. The threat of more constrictive regulation, now apparently invoked in Germany, may also be a decisive factor if it prompts a positive response from industry (see Chapter five).

Social responsibility culture

Although the social responsibility of British and German businesses was not directly assessed in the producer survey, the general impression from Chapter six was that German respondents have accepted the environmental necessity of government regulation. Such factors could conceivably be assessed in future research using indicators such as companies' involvement in EMAS and ISO 14000 environmental management programmes. The DSD's emphasis on encouraging public participation is also a clear distinguishing feature (Chapter five). This has taken three basic forms; investment in collection networks (73% of packaging waste in Germany is now recovered through kerbside or mixed collection), the symbolic association of the Green Dot with recycling, and the organisation of annual recycling-awareness days (Michaelis, 1995). By contrast, the UK Regulations have maintained a low public profile aside from occasional retailer initiatives, mostly in the form of 'green box' or 'bag for life' schemes. Whilst the DETR has sought to reinvigorate the public awareness initiatives operated by UK compliance schemes (DETR, 1998a), one of the biggest obstacles to public participation has been the failure to provide local authorities with guaranteed access to PRN funds, a situation which has stunted investment in waste collection.

Therefore, of the possible factors separating the British and German PWMSs, the most important appear to be the coercive powers held by national authorities in terms of legislation and implementing mechanisms, and the extent to which environmental initiatives galvanise public participation. However, this does not alter the fact that the German model has achieved environmental objectives largely at the expense of economic efficiency, or that in both cases environmental charges have facilitated the execution of policy implementation rather than determined its direction.

7.6.2 The 'Impotency' of Price-based Environmental Regulation

The finding that environmental charges produce only marginal changes in polluter behaviour has substantial implications for the wider use of price-based policy instruments. Whilst the notion of environmental incentive taxes has a formidable theoretical coherence, the reasons for its fragility under empirical investigation require further investigation. Two factors appear best to explain this phenomenon, the low financial impact of environmental charges and the ability of producers to diffuse the costs imposed by price-based regulation.

The Financial Impact of Environmental Charges

Prior to the survey, a number of contacts were made with UK-based manufacturers and retailers (see Chapter four). Though these data are limited and do not include opinions from German industry, their comments help to explain the absence of a relationship between environmental charges and business actions. One electronics manufacturer, for instance, stressed that its annual compliance costs amounted to £30,000 compared with a turnover of £870 million. The respondent therefore argued that the possible savings from projects re-evaluating the design and consumption of packaging did not justify the expenditures involved. Three other respondents acknowledged that the Regulations had increased the pressure for packaging re-design but felt they were overshadowed by other business considerations (such as the logistics and marketing benefits of packaging). This corroborates Jones' (1999) point that businesses rarely make major operational commitments in response to relatively minor cost pressures. Another manufacturer suggested that as the business routinely explored all opportunities for packaging 'optimisation', neither the Regulations nor economic instruments had influenced their decisions. The British Retail Consortium (BRC) submission to the 1998 review of the Regulations echoes these sentiments (BRC, 1998: 3-6):

Many retailers have long been doing all they can to encourage recycling and minimise packaging use. Examples include provision of banks on car parks; specification of recycled materials in packaging; packaging minimisation programmes; increased use of reusable packaging as demonstrated by closed-loop reusable schemes; and heavy investment in equipment for recovery of backdoor packaging waste. However, packaging is necessary in terms of product protection and health and safety considerations. Reuse will occur if there is an economic benefit to it.

A major explanation for the lack of relationship between compliance costs and business actions therefore seems to be that environmental charges are simply too low in relation to company turnover and profit to have a major impact on business behaviour. This attitude may be further reinforced by the tendency to seek 'fiscally-neutral' environmental taxes. It is frequently argued that environmental taxes should not increase the aggregate tax burden on the economy but should instead shift the balance from economic goods to environmental bads (Gee, 1997; Ekins, 1997). However, policies adopting this approach may be methodologically flawed when viewed from the perspective of corporate decision-making. If businesses are faced with new environmental taxes but concurrently rebated on their labour costs through reductions in employer's National Insurance contributions, this may dissipate much of the behaviour-changing potential of environmental charges. This argument clearly requires further development but, at face value, it may help explain why the Landfill Tax has not stanchied the increase in waste going to landfill disposal in the UK (DETR, 1999b).

Economists would argue that this is simply a question of price elasticity and that environmental charges can be raised to the point where optimal abatement incentives are created (Jacobs, 1991; Pearce and Turner, 1992). However, there are two problems with this position. First, considering the recycling costs already incurred by German industry, it may be politically unfeasible to raise environmental taxes to the point where polluting products become price-elastic (Baumol and Oates, 1988). Second, higher charges are likely to encourage more firms to disregard the law, bringing the issue of effective enforcement again to the fore (see Chapter two). A final consideration for the PRN system concerns the fact that market forces are the only mechanism determining this particular environmental charge. As recycling markets respond to a variety of influences, including the regulatory regime, producer willingness-to-pay, competitive pressures, international commodity prices and general macro-economic conditions, it is dangerous to over-stylise the relationship between environmental charges and polluter behaviour.

Cost Diffusion

Cost diffusion, as the name suggests, occurs where businesses seeking to maximise profits (or minimise losses) disperse avoidable costs through the supply chain. This can take several forms, though the most common are increases in product prices and pressure on suppliers to re-design products or offer price concessions. Although this has the benefit of disseminating the PPP to all polluting parties, it can also dissipate the incentive effect if, through dilution, environmental costs become a negligible consideration at each stage in the chain. The *overall* impact of the Regulations on the Retail Price Index has been estimated at 0.1-0.7% per annum (Daily Telegraph, 1997). The extent of cost diffusion by the two respondent groups is assessed in Table 7.7. It shows that whilst neither group intends to recoup all compliance costs from customers, German respondents favoured some price increases though, interestingly, the British group did not support this option. Although there was some indication that German firms would inform their customers of the reason for price increases, British respondents again seemed intent on keeping packaging waste costs out of the public gaze.

These results therefore suggest that as environmental charges increase, cost diffusion becomes more widespread. It was not considered feasible to interrogate respondents on the methods used to disperse environmental costs to suppliers, as this can be achieved in numerous ways. It is nonetheless likely that some amount of cost diffusion up the supply chain is taking place (Hill, 1997). Hill also proposes that market-led measures have not proved as effective as legislation in terms of exerting environmental pressures through the supply chain. Jacobs (1991) concludes, however, that the higher visibility of consumption-based taxes sets up a stronger dynamic for change than producer-related charges but concedes that the political unacceptability of direct consumer taxes is a major obstacle to this strategy. Therefore, whilst a single study cannot conclusively prove or disprove the potency of incentive taxes, polluters clearly respond to numerous market stimuli, many of which fall beyond the compass of government regulation. Price iterations may be a politically prudent way of determining the acceptable boundaries of economic instruments but there is no guarantee that the incentive effect will be reached first. Ultimately, both policy-makers and polluters are required to make fine tactical judgements in pursuit of their preferred policy outcomes (Lévêque, 1996a; Heyes, 1998).

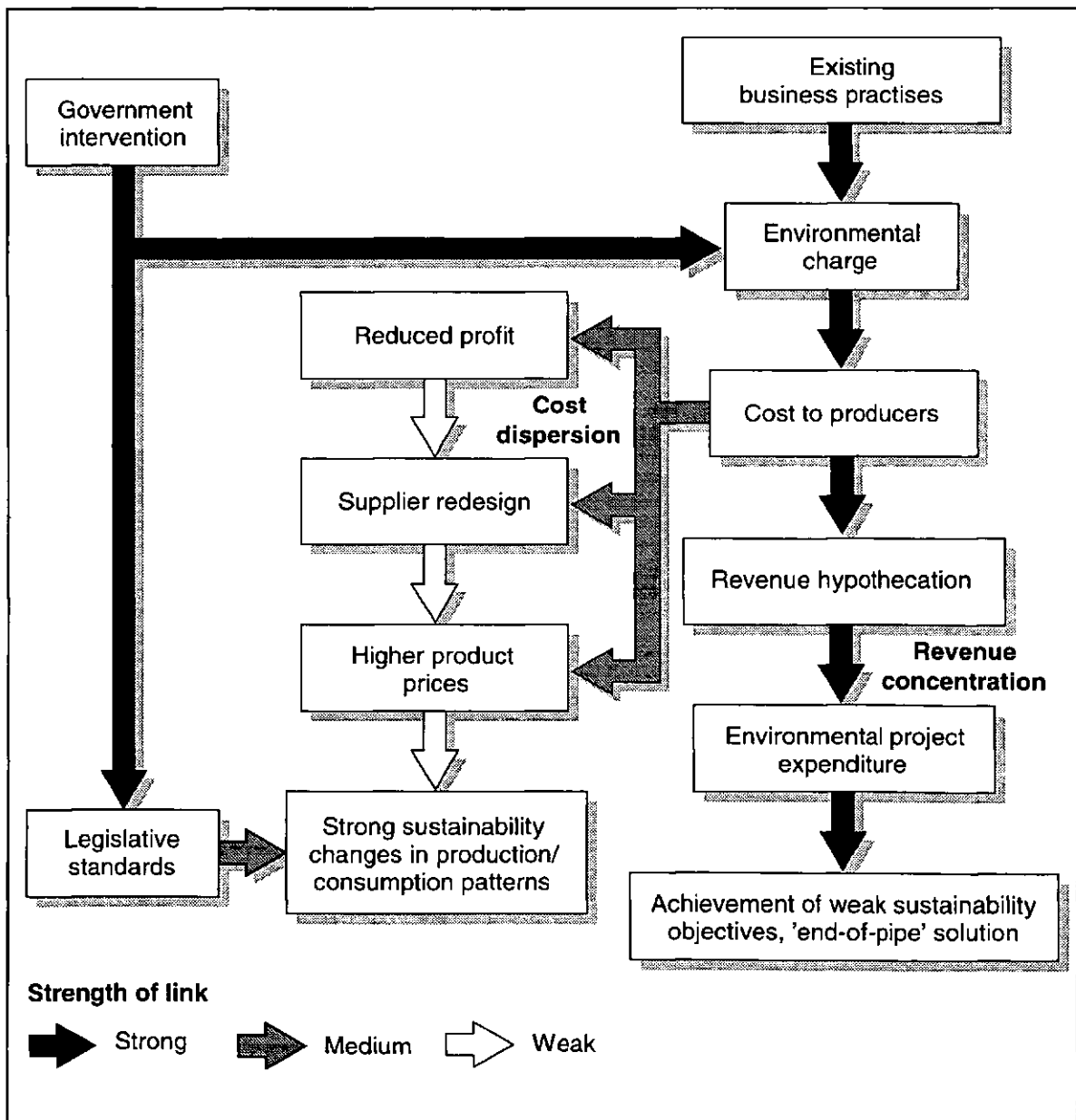
Table 7.7 Diffusion of Packaging Waste Charges

Measure		Descriptive Statistics					Mann-Whitney U		
		N	Mean	Standard error	Standard deviation	Sig. of variance from 0	Mean Rank	U statistic	2-tailed sig.
Company will increase prices to recover all compliance costs	UK	424	-0.62	0.056	1.15	0.000	288.28	32121.0	0.000
	Germany	196	-0.15	0.083	1.16	0.076	358.57		
Company will increase prices to recover part of compliance costs	UK	416	-0.26	0.059	1.20	0.000	273.93	27220.5	0.000
	Germany	195	0.46	0.084	1.18	0.000	374.41		
Company will inform customers of reasons for price increases	UK	408	-0.32	0.053	1.07	0.000	277.77	29893.5	0.000
	Germany	195	0.18	0.081	1.14	0.029	352.70		

7.6.3 A Typology of Business Responses to Environmental Policy Instruments

The final question raised by the survey concerns the sustainability benefits engendered by price-based environmental regulation. This can be answered in part by constructing a model conceptualising industry's response to economic and legislative policy instruments (Figure 7.3). It has been argued that legislative standards produce clear but usually narrowly focused environmental-improvement standards (Bohm and Russell, 1985). Moreover, their environmental efficacy depends on the enforcement capability of government agencies (Lévêque, 1995). Recognising that legislative standards integr-

Figure 7.3 Producer Responses to Environmental Policy Instruments



ate environmental concerns less comprehensively than economic instruments and that effective enforcement may be appreciably lower than nominal compliance, legislative standards have, at best, a moderate link with strong sustainability outcomes. Even this depends on the coercive power held by government and the degree of information symmetry between regulators and regulated during the preparation of pollution standards (Lévêque, 1995).

Price-based regulation, by contrast, seeks to generate abatement incentives and hypothecation revenue by re-internalising the environmental externalities created by industrial activity. Under this system, earmarked taxes can then be used, *inter alia*, for environmental expenditures. However, the evidence is that the incentive effect of environmental taxes is often minimal, first, because governments seek to avoid damaging the economy with high taxes and, second, because they are a relatively minor component of the total factor costs for price-inelastic commodities. Cost dispersion processes may then further erode the incentive effect. Though company profits may be affected slightly, the likelihood is that a proportion of any shortfall will be regained through cost savings elsewhere. Where these conditions are met, the link between price-based regulation and strong sustainability will tend to be weak.

However, the use of Green Dot and PRN revenue for investment in recycling infrastructure suggests that the hypothecation of environmental taxes has been the main benefit of the two PWMSs (see Chapter five). Regardless of the market distortions currently afflicting the PRN system, a process has been established whereby the revenue raised from many polluters has been concentrated towards abatement activity. This has funded an annual recycling programme of £1.4 billion in Germany and has made £56 million available in Britain³ (DETR, 1999a; DSD, 1999a). As a result, 64% of packaging waste produced in Germany is now recycled and, potentially, the recovery rate in the UK could achieve 50% by 2001. Nevertheless, the sustainability outcomes produced by environmental hypothecation are generally weak, as pollution must occur before revenue becomes available for investment. Expenditures are therefore targeted at 'end-of-pipe' measures rather than combating the source of pollution and even this is

³ This figure is an estimate produced from DETR data (DETR, 1998). Approximately 90% of PRN trading takes place between August and January. To derive the revenue available, therefore, average prices for this period were factored against the number of PRNs issued in 1998.

dependent on the judicious use of hypothecated revenue by its recipients⁴. In the UK, this appears to have been the main environmental downfall of the market-led system. For governments seeking to use hypothecation to pursue stronger sustainability outcomes, there is therefore a strong case not only for overseeing the flow of funding, but also for ensuring that earmarking arrangements divert a proportion of revenue towards challenging the production of the pollutant being taxed.

It should be noted, however, that both practical and theoretical objections have been lodged against the hypothecation of environmental taxes. They are seen by many as having a distortionary effect on the economy; O'Riordan (1997: 38) notes that the UK Treasury has a 'well-established and even doctrinaire opposition to earmarking because of the rigidities it introduces into taxation revenue.' Barde (1997) criticises hypothecation because it moves environmental taxes away from their original purpose of changing polluter behaviour and adds that if taxes do incidentally reach the incentive level, reductions in pollution may induce over-capacity in pollution-control facilities and, hence, economic inefficiencies (also Rajah and Smith, 1993). There is also the question of matching environmental tax revenues with expenditure requirements. Smith (1997) considers that only where this occurs naturally (in Smith's opinion, an unlikely contingency), can both be set at the correct level. Against this, the evidence from the Packaging Directive suggests that where earmarking is designed to ameliorate the problem targeted by the tax, it is possible to create a focused and closed-loop system of taxation and expenditure.

As with all models, the temptation is to over-simplify relationships in order to make complex systems more comprehensible (Hahn, 1989). It is apparent that the outcomes produced by environmental policy strategies are shaped by many inter-connected factors. This model does not purport to unravel all the intricacies of corporate responses to legislative and economic instruments but, rather, it seeks to conceptualise particular facets of industry behaviour and relate them to the policy objective of

⁴ Defensive expenditures have also been the main focus of the UK's Landfill Tax Credit Scheme and sometimes revenue has been earmarked for uses with only a peripheral link to the environmental problem being addressed. The House of Commons Select Committee on Environment, Transport and Regional Affairs (HCSCETRA) Thirteenth Report noted that although the scheme seeks to encourage more sustainable waste management practices, a disproportionate amount of revenue was being allocated to the protection or provision of public amenities in the vicinity of landfill sites compared with measures promoting re-use, recycling and end-use markets (HCSCETRA, 1999).

sustainable development. It is therefore presented primarily as a basis for discussion. However, it is clear from this analysis that the greatest gains from economic instruments have come from the generation of revenue to defray environmental expenditures rather than from their incentive effect.

As noted earlier, this is not the first study to question the environmental efficacy of price-based regulation. Hahn's (1989: 108) assessment of emission fees in France, the Netherlands and the USA, concludes that with a few exceptions:

The major motivation for implementing emission fees is to raise revenues, which are earmarked for activities which promote environmental quality ... most charges are not large enough to have a dramatic effect on the behaviour of polluters. In fact, they are not designed to have such an effect. They are relatively low and not directly related to the behaviour of individual firms or consumers ... presumably, starting out with a relatively low charge is a way of testing the political waters as well as determining whether the instrument will have the desired effect.

However, the prevention of packaging was a primary objective of the Packaging Directive and as most Member States declined to introduce legislative standards with this intent, economic incentives have been the main mechanism employed (Eichstädt *et al.*, 1999). Whilst some successes have been achieved, these have been created by legislation and 'command-and-control' measures rather than the re-internalisation of externality costs. It is nonetheless apparent that the advances in environmental protection achieved in both countries would not have been possible without the provision of funds to finance major environmental expenditures. Thus, it is apparent that economic instruments serve a very real and useful function but, if stronger sustainability is to be achieved, the development of hypothecation measures geared towards changing production and consumption patterns needs to be part of the overall policy approach.

7.7 Conclusions

Any research examining the implementation of environmental policies inevitably encounters the fact that policy-makers are required to balance competing policy objectives (Haigh, 1998). As such, environmental sustainability cannot be considered in splendid isolation from economic and social considerations. There are therefore no

perfect solutions; each policy instrument has its relative merits and limitations, and no single mechanism can achieve the social and ecological aspirations of sustainable development. The economic approach has been vaunted in some quarters as the most effective method for managing environmental problems, but the evidence from this study suggests that its potential is difficult to realise fully in practice. Furthermore, as Beder (1996: 61) notes:

If environmental degradation is indeed a result of a failure to price environmental goods and therefore harness self-interest to the common good, then economic instruments could well provide a much needed solution. However, if environmental degradation has resulted from making environmental concerns secondary to economic concerns, and having decisions made by people who see environmental resources merely as an adjunct to production, then economic instruments will merely perpetuate the problem and subvert any potential for political and value-based change.

Ten years on from the first of their *Blueprint* series, Pearce and Barbier (2000) reflect on the changes that have occurred in environmental policy since its publication. They conclude that environmental economics has become a common language for scientist and policy-makers alike and that its opponents are fewer as many have realised some virtue in the economic approach. However, they concede that putting the economic message into practice has been more difficult than anticipated, as it has necessitated changing (unsustainable) institutions that have been built up over many years. As an addendum, they warn that the focus should now be on the task of applying environmental economics rather than on reconstituting the problem. Whilst this study has not sought to oppose these conclusions, it has highlighted some of the practical difficulties (and benefits) of putting environmental economics into practice. It has presaged the dangers of over-stylising the relationship between polluter costs and behaviour in complex market situations and argued that, as a result, policy instruments must be carefully selected if both environmental and economic objectives are to be achieved. In particular, it has been suggested that the incentive effect of economic instruments may be more elusive than expected. In some cases, it may be unattainable. Hypothecation has a better practical record though, wherever possible, it needs to incorporate measures to combat the production of pollution rather than ones that merely ameliorate its effects. Even where hypothecation arrangements only produce weak sustainability outcomes, however, prudent utilisation of this approach could avert a protracted policy detour in search of the incentive effect of economic instruments.

Chapter Eight

Implications of the Packaging Directive for EU Environmental Policy

8.1 Introduction

The previous chapter presented a framework for evaluating the effectiveness of environmental policy instruments that takes account both of normative economic considerations and the political-commercial realities of their application. It proposed that if environmental charges are analysed in isolation from industry's broader and more opportunistic decision-making processes, this creates an idealised and distorted view of policy-instrument efficacy. Policy-makers must instead be aware that businesses invariably make commercial trade-offs when planning responses to price-based regulation and that normative modelling is rarely an accurate predictive tool. Recognising, therefore, that the context in which price-based environmental regulation operates must be appreciated if their functionality is to be properly evaluated (see Chapters two and three), this chapter considers how the EU's policy-making style has affected the success of economic instruments.

Chapter three identified three main characteristics of EU environmental policy. First, the EU is a unique political and judicial system with highly complex institutional and decision-making arrangements (Barnes and Barnes, 1999). Although its general aim is to promote European integration, practical policy-making involves intense negotiation between policy actors intent on protecting their national or sectoral interests. Tensions within the integration process are expressed in terms of; (i) deliberations on whether EU or Member-State action is more appropriate (guided by the subsidiarity principle), and (ii) the Commission's extensive use of directives to inject flexibility into the implementation process. The key question here, therefore, is whether the introduction of economic instruments to implement the Packaging Waste Directive has promoted greater convergence in Member-State environmental standards than occurred under 'traditional' command-and-control regulation.

The second characteristic is that progress towards legislative harmonisation and sustainable development has been hampered by the poor implementation of EU environmental law by the Member States (CEC, 1996c; 1998a; 1999b). By common

assent, the procedures to combat this under Article 169 are cumbersome and often only partially effective (Collins and Earnshaw, 1993; Demmke, 1997; Barnes and Barnes, 1999). The second question is whether price-based regulation within the framework of Member State-led implementation has improved policy implementation or whether the problems experienced under command-and-control regimes are being replicated within the price-based approach.

The final characteristic is that despite the significant benefits flowing from the Europeanisation of environmental policy, conflicts of interest - either intermittent or fundamental depending on one's perspective - exist between the objectives of the environmental programme and other EU priorities, notably economic and trade development. Again there is general agreement that the integration of environmental considerations into other policy spheres is incomplete despite it being an essential requirement of sustainable development (Blacksell, 1994; Forrester, 1999). The chapter therefore examines the extent to which the formulation and implementation of the Packaging Directive has assisted the integration of environmental concerns into other policy spheres and the management of potential conflicts. This raises the final question explored in this chapter, whether the EU's fundamental mission of economic and trade development is compatible with the tenets of sustainable development.

8.2 Convergence, Persistent Diversity or Divergence in Member State Policies?

The extent to which EU legislation has promoted the convergence of Member-State environmental policies has been extensively debated in the literature (see, for example, Aguilar Fernández, 1994; Haigh, 1994; Scott *et al.*, 1994; Majone, 1996; Lowe and Ward, 1998a). While these studies broadly agree that policy and judicial Europeanisation have profoundly influenced the environmental actions of the Member States, they acknowledge that national institutions continue to take most of the important decisions concerning the practical implementation of EU policies (Aguilar Fernández, 1994; Majone, 1996). Two aspects of convergence must therefore be considered; the formal harmonisation of legal standards and the convergence or otherwise of national policy-implementation styles. These styles reflect the prevailing institutional and planning procedures in each Member State and, in turn, influence the design of economic instruments. This section therefore explores three questions. First, in what areas are Member-State packaging policies converging or diverging? Second,

how far can these trends be attributed to general EU institutional procedures or the specific use of economic instruments by the Member States? Finally, to what extent do the EU's legislative and institutional arrangements impinge on the environmental effectiveness of economic instruments?

8.2.1 Formal Legal Convergence

Chapter five noted that EU action on packaging waste was initially prompted by the German Packaging Ordinance of 1991 and concerns amongst other Member States that their industries would be negatively influenced by the German legislation. If action was not taken by the European Commission, they feared, their businesses would be forced to compete against discriminatory regulations, particularly in relation to re-fill quotas, in order to gain access to the German market (Simmonsson, 1995). Although the Commission immediately challenged the re-fill quotas (Chapter five), there was a general acceptance that EU legislation was needed. Thus, the debate on subsidiarity appears to have been quite muted (Golub, 1996; Haverland, 1999). Discussions centred less on whether EU regulation was appropriate and more on what form it should take. As with previous EU environmental legislation, the trade argument was a critical factor. However, the prominence of the environmental problem in question swung the political pendulum in favour of harmonising legislation rather than an ECJ case against the German Ordinance. Though post-consumer waste is not a trans-national problem in the same sense as air or water pollution, international waste shipments had become particularly contentious in Belgium¹, whilst most Member States were experiencing problems managing their domestic waste disposal. Overturning the German legislation would therefore have been counter-productive both for the EU environmental programme and for national waste management strategies (Golub, 1996). In the event, many Member States welcomed the Commission initiative (Haverland, 1999).

Despite the tortuous process of policy negotiation (see Chapter five), Haverland (1999) argues that the Packaging Directive has encouraged a marked convergence in

¹ Wallonia introduced a ban on waste imports in 1992 in order to prevent the region becoming the 'dustbin of Europe.' The ECJ accepted the Belgian law even though it inhibited the free movement of waste as a good in the Single Market (Case C-2/90, *Commission vs. Belgium* [1992] 1 ECR 4431). The Court decided that the proximity principle of environmental policy meant that, wherever possible, the international movement of waste should be kept to a minimum. It also established a more general principle, that where there was a particularly acute local environmental problem, as was the case in Wallonia, some 'exceptional' exemptions to the free movement of goods should be granted (Lister, 1996).

packaging waste legislation in the Member States. Logically, once all states had accepted the Directive's final recovery and recycling standards, they were obliged to transpose and implement its provisions in full. Whilst this might seem an obvious point, it demonstrates that EU legislative standards created greater policy convergence than would have occurred without Commission intervention. Haverland (1999) considers it unlikely either that Germany would have amended its original Packaging Ordinance or that regulation would have been introduced in Britain without the impetus of EU legislation. The introduction of harmonising legislation has been shown to have a similar impact on national policies in the area of water quality legislation (Haigh, 1994; Ward, 1998) and can therefore be seen as a powerful force in promoting policy convergence. On the other hand, Chapter five argued that the inclusion of banded targets in the Directive shows that EU legislative processes are only capable of producing approximated harmonisation because of the need to accommodate the wishes and implementation capabilities of both leader and laggard states (see also Liefferink and Andersen, 1998; Knill and Lenschow, 1998).

8.2.2 Convergence in Implementation Style

Notwithstanding the convergence of formal environmental standards as a result of EU legislation, it is common practice for the Member States to adapt directives to their preferred objectives and procedures during their implementation (Weale, 1996; Lowe and Ward, 1998a). This enabled Germany to tackle the problem of packaging waste using its customary blend of precautionary and prescriptive policies and Britain to retain its arguably more pragmatic and neo-liberal stance towards environmental policy (Bailey, 1999a). It should nonetheless be recognised that the process of legal harmonisation circumscribed the actions of Member States and, therefore, the degree of permissible diversity. The fact that the UK government was forced to introduce additional regulation to the 'market-led' reprocessing sector in order to increase its chances of complying with EU targets is a clear example of such a constraint (see Chapter five)². Accordingly, where EU directives contain clear standards and implementation deadlines, the EU's institutional procedures and requirements place

² In fact, Vogel (1996) argues that free-market policies rarely lead to reduced regulation and cites the experiences of industrialised countries that have adopted neo-liberal philosophies but increased industry regulation in order to ensure that market forces do not undermine public policy objectives. A case in point in relation to rail privatisation in the UK can be found in Shaw (2000).

tangible boundaries on the leeway open to national authorities (Krämer, 1991)³. Whilst such constraints are not unique to environmental policy, this does not alter the fact that directives have acted as a powerful force in maintaining an 'approximated' cohesion in EU environmental policy despite their method-permissive character. Looking to the future, it is possible that the Commission's ambition to increase the number of framework directives containing only broad objectives and time-frames - ostensibly to simplify and make environmental policy *more* flexible - may relax these constraints (Barnes and Barnes, 1999). Though their impact on policy convergence will only become clear when more framework directives come into effect, they are nonetheless a development that requires monitoring and research.

Notwithstanding the constraints imposed by EU procedures, the implementation of the Packaging Directive in Britain and Germany has been heavily influenced by national approaches to environmental policy, as national governments have lost none of their preferences or decision-making capacity during the latter stages of the policy process (Golub, 1996; Kohler-Koch, 1996; Bailey, 1999a). Haverland (1999) cites two factors as explaining the persistent diversity of national packaging policies; one generic and one specific to the Packaging Directive. The first is institutional inertia, which occurs where Member-State governments are unwilling to abandon existing implementation structures because of the political capital invested in them (Knill and Lenschow, 1998). This was exhibited in Germany by the *Bundesrat's* opposition to policy Europeanisation in the form of weakened re-fill quotas (see Chapter five). Similarly, the federal government preferred to spend 800 million Deutschmarks baling out the DSD in 1993 rather than see the system crumble, despite its long-term viability being less assured at the time than it is currently. Even the present UK administration, which promised a sweeping review of the PRN system when in opposition, has been noticeably more cautious about reform since it came to power. Instead, following the initial swathes of packaging legislation, most EU governments have adopted an incrementalist approach to policy change and preferred to test each policy adjustment tentatively in order to avoid major expenditures or political risks⁴.

³ Krämer (1991) further notes that, in accordance with Article 177 of the Treaty, Member States cannot avoid their responsibilities to take or refrain from certain actions by failing to adopt the necessary implementing measures by the relevant deadline or in a correct manner. The ECJ ruled in Case 152/84 (FN I) Marshall, 748 that any other interpretation would enable Member States to rid directives of their direct effect simply by failing to implement them or implement them properly and would be contrary to the intentions behind the implementation flexibility of directives.

⁴ Vogel (1993b) observes that the US federal government also tends to conduct its environmental policy in this highly circumspect manner.

The second cause of persistent diversity, according to Haverland, was the high profile of packaging waste policy for some sectors of industry and the public. Whilst public opinion has not been a decisive factor in Britain because of the relative obscurity of the PRN scheme, industry's desire not to be burdened by a second and potentially contradictory tranche of regulation has dissuaded the government - which took much self-credit for its extensive consultation process prior to the UK Regulations - from making wholesale changes to the system (ACP, 1998). In Germany, the contentiousness of the Dual System made it virtually impossible for the federal government to make radical changes to the scheme without back-peddalling on its commitments to reduce packaging waste. Furthermore, its cautious approach was generally supported by German industry, which was reluctant to abandon the seven billion Deutschmarks it invested setting up the Dual System (Haverland, 1999). Thus, both governments have recognised that maintaining industry goodwill is critical for the success of their packaging policies, whilst industry's priority has been to minimise the disruptive effects of major policy shifts.

Though the full extent of policy convergence cannot be evaluated until the Directive's compliance deadline in 2001, the main processes acting in favour of convergence have been, first, Member States' agreement that packaging policy was a legitimate area for EU intervention and, second, the use of legislative standards to promote formal integration. Within this framework, however, the derogations in the Directive and the Commission's emphasis on flexible implementation have led to continued diversity in the way the Directive has been applied. Kerremans (1996) ascribes this partly to the complex nature of EU policy-making and the fact that long-term co-operation is needed between Member States in order to maintain policy consensus. Exemptions for minority groups at both ends of the spectrum are therefore commonplace even under majority voting. More importantly, the flexibility of directives enables Member States to minimise administrative costs and implement EU legislation at a pace they can manage⁵. Thus, it is in the Commission's interests to tolerate 'controlled' diversity in Member-State standards in order to maintain the support of national authorities for the overall environmental programme. The policy-making skill therefore lies in judging

⁵ Though Knill and Lenschow (1998) argue that the level of embeddedness of national administrative structures is a stronger influence on policy actions than the costs of adaptation to EU legislation.

how best to maximise the benefits of implementation flexibility whilst maintaining a reasonable coherence in the environmental *acquis* (Krämer, 1998; Temmink, 1999).

However, the evidence from the Packaging Directive indicates that economic instruments have, in fact, increased the diversity of national policies beyond that intended by the Commission. Whilst there can be no uniform 'rational' formula for calculating and apportioning pollution costs in all Member States, as different circumstances prevail in each country (Rees, 1997), the recycling charges in Europe owe more to the ideological allegiances of each national authority than they do to any objective valuation criteria. Far from harmonising the environmental costs, incentive patterns and systems of revenue hypothecation in the Member States, the flexible approach has infused existing differences deeper within national economies. Regardless of whether any system is superior to others, allowing economic instruments to be implemented at Member-State level increases the likelihood of fragmentation in the EU environmental programme. Whilst it might again be argued that this enables Member States to manage EU environmental policy according to their capabilities, uncertainties as to the incentive patterns created by economic instruments makes the boundaries to divergence less easy to discern. It should be remembered that even the UK government took two years to recognise and act against the market distortions caused by its own PRN system. The Commission and Court of Justice are therefore likely to find it more difficult to determine whether Member-State economic instruments contravene EU free-trade rules than they did to identify and challenge infringements of the Directive's legislative provisions. As the boundaries become less easy to adjudicate, maintaining the cohesion and purpose of the EU environmental programme will become an increasingly stern challenge (Bailey, 1999a).

Another example of where EU institutional procedures have caused policy divergence can arguably be found in the case of the Commission's carbon/energy tax initiative. Although carbon taxes were the subject of extensive studies in the early 1990s, little progress was made towards their introduction on an EU-wide basis (CEC, 1992b; 1997c). The principal reason for this was that the Council automatically employs unanimous voting for all taxation issues, enabling Member States to invoke Article 93 and block EU legislation that conflicts with their national interests. As any EU-wide carbon tax initiative is likely to impact upon state taxes, subsidies and industry competitiveness, it was almost guaranteed that the proposal would founder on national

objections (Barnes and Barnes, 1999). Both Britain and France fought what they saw as a fiscally doctrinaire carbon tax, whilst the initiative was also opposed by the powerful energy-intensive industry lobby (Long, 1998).

Table 8.1 Carbon and Energy Taxes in the EU

	Austria	Belgium	Denmark	Finland	France	Germany	Greece	Ireland	Italy	Luxembourg	Netherlands	Portugal	Spain	Sweden	United Kingdom
<i>Motor Fuels</i>															
Leaded/unleaded (differential)	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
Diesel (quality differential)			•	•	•									•	•
Carbon/energy taxation			•	•	•						•			•	•
Sulphur tax			•	•	•								◦	•	•
Other excise tax (non-VAT)	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
<i>Other Energy Products</i>															
Other excise taxes	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
Carbon/energy taxation	•		•	•							•			•	◦
Sulphur tax			•		•				•				◦	•	•
NO _x charge					•			•					◦	•	

☼ From April 2001

◦ Only in the autonomous region of Galicia

Source: Barde (1997: 228-9, updated)

Despite these objections, the EU was keen to put measures in place which demonstrated its global leadership under the Kyoto Protocol on climate change (Zito, 2000). The Commission proposal for an EU-wide carbon tax was therefore replaced by an agreement allowing Member States to introduce national taxes on energy consumption and emissions. This has led to a range of national initiatives, including the Danish and Dutch carbon and energy taxes and, from 2001, similar measures in the UK (see Table 8.1). Although there is considerable overlap in national energy taxes - for example, all countries have differentiated duties on different types of motor-vehicle fuels - Barnes and Barnes (1999: 147) allege that the only common energy tax with a

noticeable effect on consumption is the excise duty on mineral oils⁶. More conspicuous are the variations in the scope and scale of national taxes (see Table 8.2). Sweden, Denmark and Finland have led the move towards incentive taxes whilst, for differing reasons, Austria, Portugal, Italy and Germany have largely eschewed them. However, provided national taxes do not impinge on inter-state trade in the EU - a point enlarged upon shortly - the Member-State-led approach does little to prevent greater divergence in national environmental tax regimes or, consequently, environmental standards. On the premise that the EU possesses neither the democratic legitimacy nor the practical means to interfere in the fiscal affairs of the Member States, it seems that environmental taxes are unlikely to encourage greater convergence in national environmental incentives and standards.

Table 8.2 Environmental Taxes on Energy Products (ECUs per unit, 1998)

	<i>gas oil</i> (kl)	<i>LPG</i> (tonne)	<i>Kerosene</i> (kl)	<i>Coal</i> (tonne)	<i>natural gas</i> (m ³)	<i>electricity</i> (%)	<i>Unleaded petrol</i> (kl)
France	40.6	25.4	-	-	0.018	-	497.5
Germany	77.1	39.1	22.0	-	-	8.5	580.6
Netherlands ^a	42.4	-	37.7	-	-	-	660.7
UK	58.5	14.9	58.5	10.5	0.0097	-	564.5

a - includes excise and environmental taxes on all fuel products.

Source: Eco-Tax Database of Forum for the Future at Keele University

8.3 Price-based Regulation and the Implementation of EU Policy

8.3.1 Introduction

Although the Packaging Directive has become a large-scale experiment in price-based environmental regulation, many aspects of its implementation follow the classic pattern of EU environmental policy (see Lister, 1996; Lowe and Ward, 1998a; Barnes and Barnes, 1999). Following protracted negotiations and consensual bargaining, the

⁶ It is doubtful whether fuel taxes could have a major impact on consumption because fuel products are generally highly price inelastic (see Chapters two and three).

Council of Ministers agreed a modified Commission proposal and Member States have, in the main, transposed and begun to implement its requirements. The Commission has resolved most minor transposition failures, leaving only those where there are fundamental clashes on whether particular methods of environmental management contravene the EU Treaty (in this case, the German and Danish re-fill provisions). As often occurs, leader states which originally pressed for the adoption of their national measures in EU legislation have already met the Directive's targets, whilst some of the more laggardly are struggling to adapt to EU requirements. In many respects, therefore, little has changed from the customary pattern of implementing EU environmental law.

The Commission and Court of Justice will ultimately judge the implementation of the Packaging Directive on the basis of whether Member States have met the required recovery and recycling standards. The Member States have generally used environmental charges in two ways to achieve these targets and the Directive's 'Essential Requirements', first, as a means of facilitating investment in recycling infrastructure and, second, to encourage industry to reduce, re-use and recycle its packaging waste (see Chapters five and seven). The question, therefore, is whether the economic instruments applied by the German and British authorities have helped to achieve either objective more efficiently than straightforward legislation?

8.3.2 EU Policy, Price-based Regulation and Infrastructure Development

Chapter five argued that the hypothecation of environmental charges for infrastructure development in Britain and Germany had produced measurable effects, but acknowledged that the management of this revenue has varied markedly in the two states. However, the use of economic instruments for this purpose was not inevitable and only followed careful deliberation by both governments. All environmental regulation, whether command-and-control or price-based, requires polluters to undertake or refrain from certain activities and, thus, compliance costs are always incurred (Jacobs, 1991; Goddard, 1995). Neo-liberal theorists argue that industry should be left to determine the methods used to achieve government targets even if the nature of the environmental problem in question makes government involvement unavoidable (Friedman, 1962; Barrett *et al.*, 1997). Whilst the Packaging Ordinance and the Dual System exhibit few *laissez-faire* influences, even the PRN scheme

increases rather than restricts the state's options for intervening in the market mechanism. However, the UK government's view was that anything short of a mandatory pricing instrument would fail to meet EU targets, not necessarily because of any industry disingenuity, but simply because the scale of co-ordination required to establish collection, sorting and reprocessing networks necessitated a formal financing mechanism (Nunan, 1999). As the packaging problem was the result of market failure, introducing a regulated economic instrument provided greater assurances that the necessary redistribution of funds would take place (Pearce and Turner, 1992). Similar logic persuaded the German government that industry needed to be coerced into devising a workable system of packaging waste management (see Chapter five). In both cases this led to industry's agreement to environmental charges (as evidenced by the PRG's original proposal cited in Chapter five), the principal benefit of which has been to finance waste recovery and recycling. There has therefore been a well-defined link between tax hypothecation and the achievement of legislative standards.

8.3.3 EU Policy, Price-based Regulation and the Incentive Effect

It was demonstrated in Chapter seven that economic instruments have been only partially effective in terms of changing polluter behaviour. Some commentators go further, maintaining that the Packaging Directive has failed to increase recycling above the levels which would have been achieved without EU legislation (see Chapter five). Whilst such claims are highly speculative and can be equally levelled at economic instruments and legislative standards, the evidence suggests that recycling rates were already increasing in Germany prior to the Packaging Ordinance (Staudt, 1997; Eichstädt et al., 1999) and that it is faltering in the UK in spite of, or even because of, the PRN system (DETR, 1998a; 1999a).

Notwithstanding this, the European Commission recently expressed concerns to the International Solid Waste Management Association (ISWMA), first, that Member-State recovery and recycling standards were diverging beyond the derogations provided for in the Directive and, second, that some countries would fail to meet EU targets (Cooper, 2000). Although it was argued that this disparity is primarily the result of factors such as the stringency and length of time national legislation has been in place, the influence of economic instruments should not be entirely discounted. The economic inefficiencies created by the German government's decision to veer towards

the precautionary principle and punitive pollution charges have not prevented industry from meeting EU requirements in the short term (Michaelis, 1995; Staudt, 1997). It might transpire that this approach causes longer-term harm to the German economy (Staudt, 1997) though, against this, the government's environmental zeal has stimulated innovation and arguably given German industry a competitive edge in European environmental markets (Beuermann and Burdick, 1998; Ostermann and Schmidt, 1998)). Nonetheless, the German approach established a clearer association between pollution and the financial costs incurred by polluters. By contrast, Britain's decision to take a cost-minimisation approach is seemingly destined to cause its default against EU recycling targets.

Goddard (1995) argues that the failure of price-based instruments to achieve policy ambitions is more often the result of a failure to appreciate how market mechanisms work than outright market failure. He contends that 'free' markets should not be blamed for profit maximisation but recognises that regulation is required in order to counteract the externality effects of market actions. As the incentive effect has proven largely elusive, economic instruments have apparently failed to achieve greater changes in industry behaviour than legislation standards. However, the hypothecation of revenue for pollution prevention (Chapter seven) has partly overcome this obstacle. Moreover, the British and German experiments with environmental taxes reinforces the conclusion that national policy preferences have a profound influence on environmental-policy outcomes and therefore their effects need to be better understood.

8.4 The European Union and Sustainable Development

8.4.1 Free Trade and Environmental Protection

The relationship between EU free trade and national environmental standards is one of the most intricate and perplexing facets of the EU environmental programme. The issue is dealt with in the EU treaties in a legally consistent, but arguably precarious, manner. Since the Amsterdam Treaty, Articles 94 and 95 (governing Member-state actions in relation to the Single Market) only become relevant to environmental issues where national legislation creates technical barriers to the free movement of goods (Lister, 1996). This meant there was an overpowering case for introducing the Packaging Directive under the former Article 100a, as packaging by definition accompanies the goods whose free movement the Treaty protects. Articles 130r and

130t (amended to Article 175 at Amsterdam) further clarified the balance between the two prerogatives. These established the Commission's right to introduce high environmental standards across the EU but permitted Member States to introduce stricter legislation provided it remained compatible with other aspects of the Treaty (Article 130r(4)) (Hughes, 1996). As these articles overlap in jurisdiction where national actions promote environmental protection but restrict free trade, the Commission and ECJ are frequently called upon to interpret which Article holds precedent within specific cases. The landmark judgements in this area - the *Cassis de Dijon* ruling in 1979⁷, and the Danish bottles and Wallonian waste ban cases - each defended the free movement of goods but permitted limited restrictions on trade where there is a pressing environmental rationale and an absence of EU legislation to regulate the issue satisfactorily.

Previous sections have suggested that the use of economic instruments by Member States will make adjudication on national provisions increasingly complex. However, most systems of environmental taxes used to implement the Packaging Directive have not resulted in trade infringements. For example, although Germany exacts higher environmental charges than the UK, both schemes exclude exported packaging but include imports. Neither therefore inhibits the free movement of goods or the contestability of their markets, as German businesses exporting to Britain are only obliged to pay environmental charges comparable to those levied on their UK counterparts and vice versa. Furthermore, as the physical reclamation of packaging waste is largely managed by recycling organisations rather than individual businesses, the technical trade barriers in this respect have proved negligible. Whilst the threatened deposit-refund charges in Germany demonstrate that economic instruments can potentially discriminate against imported goods, the major trade infringements to date have not been caused by price-based provisions.

However, it might be argued that the trade-neutrality of environmental taxes could have considerable implications for the 'push-pull' dynamic of EU environmental policy (Sbragia, 1996). The push-pull dynamic, it will be recalled from Chapter three, occurs where an environmental 'leader' state introduces legislation which threatens EU free

⁷ The *Cassis de Dijon* ruling (Case 120/78, OJ C 256, 3 October 1980) established that products *must* be allowed access to markets throughout the EU provided they are safe and pose no threat to the consumer. The ruling, which reversed the previous onus on this matter, therefore requires national authorities to demonstrate *why* another Member State's standards do not adequately protect its citizens.

trade and the Commission, in order to ward off this threat, proposes EU-wide legislation. Proponents of flexible environmental policy routinely use the push-pull argument as a basis for supporting the devolution of environmental policy, claiming that the actions of leader states encourage higher environmental standards across the EU. However, if national taxes remain compatible with the free-trade elements of the EU Treaty, the pressure for harmonisation may also diminish. This has arguably already been observed in relation to fuel duties and carbon/energy taxes in the Member States. Whilst there is still likely to be pressure to harmonise the legislative standards which usually accompany environmental taxes, the effects of an environmental tax system are manifestly more far-reaching than individual legislative provisions, such as the German re-fill quota. In simple terms, the abandonment of re-fill quotas would not precipitate the complete dissolution of the Dual System. However, the adjustment of an environmental tax could fundamentally alter the incentive patterns for industry and have wide-ranging environmental implications. This does not necessarily mean that 'EU-compliant' taxes will completely negate the push-pull dynamic - this is probably far from true - but rather that a growing emphasis on price-based regulation could reduce the dynamic as a source of upward pressure on EU environmental standards.

8.4.2 The Packaging Directive, Integration and Sustainable Development

If, as the previous section indicated, the use of economic instruments in the Packaging Waste Directive has created few implications for the Single Market aside from accentuating differences in national environmental standards, what has the case study revealed about the ability of EU environmental policy to promote sustainable development? Chapter three signalled that the integration of environmental criteria into policy decisions is a critical component of sustainable development and one that the Commission has repeatedly emphasised in the EAPs (CEC, 1992a, 1994; Baker 1997). At face value, this pledge makes a straightforward connection between environmental integration and the achievement of sustainability. However, Chapter two established that sustainability and sustainable development encompass a range of perspectives regarding the relationship between society and nature, as well as the interchangeability of natural and human capital. If the question is simply whether economic instruments have integrated environmental concerns into the wider community, clearly they have internalised the environmental costs of packaging waste for over 28,000 business in Germany and Britain. However, their failure to influence industry behaviour suggests

that the *existence* of integration alone is insufficient to promote sustainability. Whilst integration is clearly necessary, there is also an obvious sufficiency requirement. Consequently, it is not satisfactory to analyse EU environmental integration as a straightforward causal process leading to a simple notion of sustainability. The more relevant question must therefore be what *form* of sustainability are the EU's institutional procedures and ideational ambitions encouraging?

8.4.3 The Obstacles to Environmentally-sustainable Policies

Musschenga (1994: 165) argues that there are five main issues which policy-makers must resolve if they are to succeed in creating sustainable environmental policies. The first is the acceptance that environmental problems exist, or the awareness problem. Whilst it has taken many years to overcome environmental 'ignorance', Musschenga concedes that the overpowering evidence of ecosystem damage, both global and local, has convinced policy-makers that something must be done (Ekins, 1993; O'Riordan and Jäger, 1995; Pezzoli, 1997). The raft of international conferences from Stockholm to Kyoto is further evidence that environmental issues are being taken increasingly seriously by the world's policy community. However, the awareness pre-condition is compounded by what Musschenga describes as the uncertainty problem, or the lack of agreement as to the seriousness of environmental problems and the best ways to approach them. Because the impact and extent of many forms of environmental degradation are still poorly understood, there are few clear guidelines on which to base action. Rational choices are therefore often impossible (also Costanza, 1993). Furthermore, where vested interests exploit this indecision, the uncertainty problem may encourage institutional inertia and lead to inadequate or inappropriate action being taken (Böhmer-Christiansen, 1994).

The third concern is the motivation problem. If one accepts that effective approaches to environmental problems will necessitate high expenditures and changes in lifestyle, many entailing a decline in certain aspects of human welfare, the question is whether people will be willing to make such sacrifices. To borrow the economics parlance, governments confronting the motivation problem must decide the extent to which human capital should be foregone in order to conserve natural capital (Pearce *et al.*, 1989). This is further complicated by the democracy problem, which Musschenga describes as the difficulties in gaining political agreement on the most appropriate

measures to be taken. The impediments to international agreement were particularly well illustrated in the interest-led wrangling which accompanied the Rio Earth Summit in 1992 (see Chapter two). Musschenga further proposes that those imposing restrictions on lifestyle risk immediate punishment at the ballot box and therefore questions whether democratic systems are capable of delivering measures to secure what are extremely ill-defined 'greater' public goods (see also Lélé, 1991). Finally, there is the justification problem. Radical measures to combat environmental problems will inevitably lead to some limitations on individual freedoms. If actions are to be based around the primacy of individual liberty, as they surely must be in liberal democracies, every limitation must be defended by an appeal to principles which justifiably over-ride that of personal freedom (Rawls, 1972).

Although Musschenga separates the environmental problem into various strands in order to understand the issues politicians must resolve, many of them are closely inter-linked. For example, the idea that policy-makers may not be re-elected if they initiate radical environmental programmes links the democracy problem to the awareness and motivation problems. In simple terms, one cannot *assume* that the general public has the same appreciation of environmental problems as policy-makers or that they are prepared to accept immediate personal sacrifices in order to promote sustainability. This is the essence of Pearce *et al's* (1989) problem of future discounting (see Chapter two). Musschenga's conceptualisation nevertheless achieves two useful objectives. First, it highlights the fundamental dilemmas faced by policy-makers in relation to the environmental problem (see also Jacobs, 1994). Second, it provides a useful framework for analysing EU environmental policy because it identifies that sustainability is likely to be achieved in evolutionary stages rather than through a seismic re-orienting of society. Thus, it has a practical focus and incorporates O'Riordan and Voisey's (1998) notion of the sustainability transition. The framework can therefore be used to evaluate the EU's progress towards sustainable development by exploring the level to which it has surmounted each obstacle.

8.4.4 The EU's response to the Obstacles to Environmental Sustainability

In addressing the awareness problem, Chapter three argued that the EU environmental programme is at least partly a well-intentioned response to the concerns expressed at the Stockholm and subsequent conferences. Furthermore, EU environmental policy has

progressively moved beyond its emphasis on free trade as a justification for policy intervention towards a broader appreciation of environmental problems. As Baker (1997: 92) comments:

Historically the Union has based its environmental protection policy not so much on a belief in the legitimacy of environmental protection as such but rather on the assumption that environmental protection measures have economic and, particularly, trade consequences. Yet despite the centrality of economic growth a new, albeit subordinate, imperative of environmental protection did evolve.

Against this, Baker (1993) reminds us that the EU has struggled to find a formulation of sustainability that is compatible with its other policy objectives. This was clearly illustrated in the Maastricht Treaty, where 'sustainable growth', 'sustainable development' and 'sustainable progress' were used as if they were interchangeable concepts. She argues that this inconsistency could not have emerged accidentally, as the Treaty was the product of protracted and politically sensitive bargaining wherein such metamorphoses in terminology could not have occurred by chance. In fact, even these manifestations were the result of intense lobbying against the Treaty's original formulation, 'sustainable growth' (Verhoeve *et al.*, 1992).

Notwithstanding this, the EU's commitment to sustainable development in the Fifth EAP demonstrates its increasing acceptance of the environmental problem. This recognition is further evidenced in the Packaging Directive by the Commission's decision to propose EU legislation rather than challenge the German Packaging Ordinance. It is less certain, however, whether economic instruments have helped to solve the awareness problem, particularly in relation to *public* awareness. Rather, the price signals sent by the PRN system have failed to conduce a significant change in public behaviour (O'Doherty and Bailey, 2000), whilst the decision by most UK businesses not to increase product prices has done little to increase public awareness (see Chapter six). Even in Germany, devices such as the National Recycling Days and the symbolism of the Green Dot have played at least as great an educative role as economic instruments. Nonetheless, the increasing number of publicity initiatives on waste and other environmental issues indicates that national policy-makers are keen to overcome the public awareness problem.

The prominence of the precautionary principle in the EAPs suggests that the EU has also taken steps to confront the uncertainty problem. Costanza (1993) notes that even in the absence of firm scientific evidence, the adoption of precautionary and preventative policies can reduce the risk of irreversible environmental damage. Yet the evidence from the Packaging Directive suggests that the EU and its Member States are more inclined to enunciate precautionary principles than they are to practice them. This is illustrated in the Directive's 'Essential Requirement' on waste prevention. Here Member States are required to ensure that:

Packaging shall be so manufactured that the packaging volume and weight be limited to the minimum adequate amount to maintain the necessary level of safety, hygiene and acceptance for the packed product and for the consumer. (OJEC, 1994: Annex II)

But that:

Member States shall [from June 2001] *presume* compliance with all essential requirements set out in this Directive including Annex II in the case of packaging which complies with [the relevant harmonised and national standards]. (OJEC, 1994: 15, emphasis added)

Thus, although undertakings on waste prevention were accepted in principle and phrased in a manner which implied that action should be taken, no tangible commitments were agreed at the EU to achieve this aim (Bailey, 1999b). In their place was a supposition that reductions in packaging waste would follow naturally from a vague stipulation that excessive packaging should not be used. The only direct actions to date in the UK have been the voluntary code of practice published by INCPEN (INCPEN, 1998) and the suggestion that PRN charges might encourage source reduction (see Chapter five). The Commission will shortly release revised targets for the Directive covering the period 2001-2006. These are expected to increase recovery standards to as high as 90% but will again refrain from introducing binding measures on waste reduction (DETR, 1999a).

However, it is possible to justify this approach on purely practical grounds rather than condemning it as an outright sustainability 'deficit'. Many states have little history of recycling, let alone waste prevention, and therefore binding reduction targets would probably be an unrealistic goal for some. Furthermore, the decision to experiment with economic instruments has enabled national authorities to explore how far the technique can be used to achieve multiple policy aims. Yet this does not alter the fact that EU

policy failed to make specific commitments to limit the production of packaging waste and is therefore only precautionary when viewed in the context of the weak sustainability model (see Chapter two). This was not a policy oversight but, rather, is symptomatic of the EU's ambiguous early forays into sustainable development and leanings towards ecological modernisation (the theory that economic development can be de-coupled from environmental degradation through environmental management processes (Chapter three)). Even so, the success of the weak sustainability approach is logically dependent on there being no deep-seated incongruity between increased economic activity and acceptable levels of environmental quality (Welford, 1999). Redclift (1996; 1997) argues that policy-makers' belief that perpetual economic expansion can be reconciled with environmental sustainability is one of the fundamental contradictions of sustainable development⁸.

In the final analysis, interpreting the precautionary principle is fraught with difficulties for the simple reason that precautionary assessments are necessary because scientific knowledge is rarely conclusive. This lack of knowledge makes it difficult to judge whether policies are merely precautionary or wholly unnecessary. Such uncertainties have prompted some Member States, including Britain, only to favour action based on firm scientific evidence (Lowe and Ward, 1998a). Moreover, as social and economic priorities must also be considered by EU decision-makers, there can be few instances where the precautionary principle does not entail a subjective evaluation of priorities. The case of the Packaging Directive nonetheless emphasises that uncertainties as to the extent of environmental problems can create a tendency towards incremental planning rather than actions to promote stronger sustainability (Weale, 1996). The situation is, of course, again exacerbated by the concurrent majorities decision-making procedure of the EU. If one accepts that economic instruments are only capable of producing weak sustainability and are dependent on legislative standards for their 'regulatory bite' (see Chapter seven), price-based intervention is unlikely to change this situation.

The final three hurdles, the motivation, democracy and justification problems, appear the most intractable within the EU's current political structure. Chapter three argued that the EU environmental programme involves an embedded struggle between the desire to promote improved standards of living and the realisation that this must be

⁸ Although policy-makers have been heavily influenced by economists, such as Beckerman (1974) and Pearce and Barbier (2000), who defend strategies that combine sustainability and economic expansion.

accompanied by responsible environmental management. This is embodied in the multiple objectives contained in Article Two of the Amsterdam Treaty:

The Community shall have as its task, by establishing a common market and an economic and monetary union and by implementing common policies or activities ... to promote throughout the Community a harmonious, balanced and sustainable development of economic activities, a high level of employment and of social protection ... sustainable and noninflationary growth, a high degree of competitiveness and convergence of economic performance, a high level of protection and improvement of the quality of the environment, the raising of the standard of living and quality of life, and economic and social cohesion and solidarity among Member States.
(<http://europa.eu.int/eur-lex/en/treaties/index.html>)

The Article demonstrates the many strands of the motivation problem for EU decision-makers. In order to achieve these multiple objectives, EU decision-makers must somehow reconcile social, economic and trade objectives with those of sustainable development. In practice, because the former constitute the Union's key *raison d'être* and more immediately affect people's lives, they frequently take precedence in planning decisions (Barnes and Barnes, 1999). Whilst most national decision-makers agree that the transition to sustainability is a major undertaking (Ballard, 2000), Welford (1999: 1) suggests that most governments have made little real effort to reverse the degradation caused by economic globalisation. He adds that policy-makers have generally clung to traditional economic indicators despite Pearce *et al's* (1989) argument that quality of life - including environmental considerations and distributional equity - should become the principal benchmark of society. The reason for this, Robertson (2000) argues, is that it is difficult for policy-makers to justify the long-term expenditures and losses in immediate material welfare which accompany sustainable development. Robertson nonetheless argues that shifting to a sustainable path will necessitate the fundamental restructuring of political and financial systems and the development of a more unified environmental agenda.

Although Musschenga's democracy problem is common to all political groupings, it is complicated in the EU by the fact that decision-making and policy implementation are shared between numerous national and EU institutions. Achieving any form of 'overlapping consensus' therefore involves intense bargaining between competing national and institutional interests (Jacobs, 1994: 162) (see Chapter three)). Whilst there is an obvious dislocation between this mode of decision-making and the form Musschenga suggests is necessary to resolve the sustainability problem, the EU cannot

simply over-ride democratic politics in the name of 'efficient' decision-making (Zito, 1998; Höreth, 1999). Furthermore, there is little evidence that the USA, with its stronger federal centre, has been willing to relinquish the democratic checks that inter-state bargaining brings to environmental policy-making (Howe, 1996). Even if a radical accord on environmental issues were possible, Musschenga's democracy problem suggests that its supporters would become susceptible to electoral rejection, especially if the measures imposed restricted immediate welfare. Therefore, in assessing how EU and national institutions might circumvent the democracy problem, the focus of attention must eventually return to the wishes of their electorates. As Jacobs (1994: 163) notes, 'If we were not to survive the environmental crisis, this is not liberal democracy's fault but our own.'

The justification problem is arguably the greatest dilemma facing EU environmental policy. For obvious reasons, democratic principles and the protection of individual liberty are absolute cornerstones of the Union. Any infringements of personal liberties must therefore be referenced against an overlapping consensus that they are necessary to combat common threats. Clearly, where only hesitant steps have been made to address the motivation and democracy problems, this justification has yet to materialise. Hession (1998) argues that liberal models of governance have become pre-eminent in modern democracies because they offer the benefits of state stability while preventing the rise of totalitarian monoliths. Rawls (1972) amplifies this point when he highlights the duties of the individual in a 'free society' (howsoever defined):

From the standpoint of natural justice, the most important duty is that to support and to further just institutions. This duty has two parts: first, we are to comply with and do our share in just institutions when they exist and apply to us; and second, we are to assist in the establishment of just arrangements when they do not exist, at least when this can be done with little cost to ourselves (Rawls, 1972: 332).

The problems come when conflicts arise between natural duties. On this point, Rawls observes:

[Though the reasons favouring the adoption of any natural duty are fairly obvious] the real difficulty lies in their more detailed specification and with questions of priority: how are these duties to be managed when they come into conflict ... There are no obvious rules for settling these questions. We cannot say, for example, that duties are lexically prior ... Nor can we simply invoke the utilitarian principle to set things straight. I do not know ... whether a systematic solution formulating useful and practicable rules is possible (Rawls, 1972: 339-40).

Whilst sustainability might readily be conceived as a just institution, making it a natural duty to defend or introduce it, policies aiming towards stronger sustainability are likely to impinge on the natural duty to defend individual liberty and welfare. Even assuming Member States accept this challenge, this is a key reason why they have experienced such difficulties integrating sustainability alongside better-established 'natural' priorities. However, it is difficult to envisage how stronger sustainability might be promoted within a system that does not possess the necessary attributes or mandate to undertake the associated sacrifices. Coates *et al.* (1997: 256-7), reviewing the role political paradigms play in resolving spatial and economic inequalities could equally have been discussing capitalism's attempts to resolve environmental crises.

The root causes of spatial inequalities cannot be tackled by spatial policies alone ... Inequalities are products of social and economic structures, of which capitalism in its many guises is the predominant example. Certainly inequalities can be alleviated by spatial policies ... but alleviation is not cure: whilst capitalism reigns, however, remedial social action may be the best that is possible ... the solution of inequalities must be sought in the restructuring of societies.

If a paradigm shift towards sustainability values were to take place - and realistically this would *have* to be an evolutionary process - policy Europeanisation and economic instruments would have major roles to play in advancing the sustainability agenda and correcting market distortions. But, at present, the fragmented and, arguably, contradictory nature of the EU policy agenda means that the environmental programme tends towards amelioration rather than cure. Whilst there is obviously an element of truth in the Commission's conviction that the Member States' failure to implement existing policies is largely to blame for this shortfall (CEC, 1992a; 1999b), the viewpoint treats environmental degradation as primarily a *technical* problem. If this means that the EU institutions fail to deal realistically with the *values* problems surrounding economic development and sustainability, then policy Europeanisation, subsidiarity and economic instruments cannot fulfil their potential. As O'Brien and Penna (1997: 185 and 197) argue:

The SEAP is a 'redrafting of Rio' in the context of ... processes of regulation and integration rather than a systematic rearrangement of society-economic-environment relationship. The contested economic, social and political institutions charged with sustaining the environmental agenda are the same institutions whose political dynamics are dissolving it.

According to this thinking, neither adjustments of responsibilities up or down the governmental ladder nor the most innovative of policy instruments will stimulate significant change. At their worst, extended discussions on the institutional issues of trans-national governance will divert attention away from a more productive debate on the substantive and pressing shortfalls in the implementation and enforcement of EU environmental policy (Demmke, 1997). Furthermore, where economic globalisation and the involvement of corporate interest groups in policy decisions act as further barriers to sustainability, defensive management and a gradual transition in values are the most that can result from the current political order. However, the alternative is 'green dictatorship'. The aim is not to advocate the latter, but rather to recognise the limitations of the present institutional and ideational structures.

8.4.5 Policy Alternatives within the present Paradigm

On a less pessimistic note, Barnes and Barnes (1999) take a more immediately practical perspective. They suggest that, in fact, EU environmental policy is not in need of a radical overhaul, as the main measures needed for effective environmental management are already in place. They add that environmental taxes and other 'new' policy instruments have much potential and warrant further exploration (see also Bailey, 1999b). Rather, they return to the criticism that national authorities have failed to implement and enforce the EU programme properly. Only once these issues are addressed, they contend, will it be possible to determine whether the existing measures are a sufficient response to the sustainability challenge. This perspective corresponds to the Commission's view in that it ascribes the resolution of the motivation, democracy and justification problems to the Member States and suggests that the faithful implementation of existing policies will promote sustainable development. Although this analysis of outcomes may be slightly optimistic, its predictions and apportionment of responsibilities are more consistent with the existing balance of power and policy options than ones which demand a radical transformation of society's values (Golub, 1997).

Probably one of the most significant recent developments in enforcement is the introduction of fines for Member States which defy ECJ rulings (Article 171). The Commission has recommended seven cases where it believes Member States should receive daily fines for persistent infringements of EU environmental law (see Table

8.3). The ECJ imposed the first actual fine on Greece for a second breach of an ECJ ruling in July 2000, and the Commission plans to initiate similar proceedings against Germany and Britain in the near future (ENDS, 2000c)⁹.

Table 8.3 **Examples of Requests for Penalty Payments to the end of 1997**

<i>Member State</i>	<i>Subject</i>	<i>Penalty payment (ECU/day)</i>	<i>Date of decision</i>	<i>Settled</i>
Italy	Radiological protection	159,300	29 January 1997	Yes
Italy	Waste management plan	123,900	29 January 1997	Yes
Germany	Surface water	158,400	29 January 1997	No
Germany	Wild birds	26,400	29 January 1997	Yes
Germany	Groundwater	264,000	29 January 1997	Yes
Belgium	Wild birds	7,750	10 December 1997	Yes
Greece	Waste water management	24,600	26 June 1997	No

Source: CEC (1998a: annex III)

In the absence of some idealistic mass realignment of Member-State values, substantive increases in the EU's enforcement capabilities is the most immediate priority. Further initiatives might include an expanded role for the European Environment Agency or, more ambitiously, the harmonisation of Member State carbon/energy taxes (Barde, 1997). The prospects for future policy co-ordination and enforcement are discussed further in Chapter nine. Though these actions still fall short of Costanza's (1989) call for policies which take account of environmental uncertainties (promoting the precautionary principle), evidence from the Packaging Directive suggests that economic instruments backed by regulatory powers have the potential to encourage significant changes if they are sufficiently well-referenced against environmental criteria. The question remains, however, whether the EU's liberal-democratic,

⁹ These actions are for failure to implement the Bathing Water and Environmental Impact Assessment directives respectively and follow initial ECJ rulings against the two Member States.

capitalist model of governance can adapt to the shifts in attitudes and regulatory disciplines that accompany sustainable development.

8.5 Conclusions

In drawing together the themes explored in this chapter, caution is required on two fronts. First, it is essential to avoid drawing expansive conclusions on the evidence of a single study. Whilst the Packaging Directive provides important lessons for future price-based environmental policies, the study's key findings require further corroboration and the ideas presented need to be developed through future research. The second is to avoid the misapprehension that there are simple answers to the paradoxes of EU environmental policy. Three main arguments have been proposed in this chapter. First, it has been suggested that the Packaging Directive has encouraged some policy convergence in the Member States but that economic instruments have the potential to encourage the greater fragmentation of environmental outcomes. Second, economic instruments have been only slightly more successful than legislative standards as a method for implementing EU environmental policy. Finally, it has been argued that the EU must overcome a series of fundamental ethical and moral dilemmas if its policies are to achieve sustainable development. At the same time, the intention has been to steer clear of trite solutions. Whilst it appears *logical* that the EU's system of liberal democracy and consumer capitalism is ill-equipped to secure more than weak forms of sustainability, there is no *practical* evidence that dictatorships, however 'green' and benign, would be more successful in dealing with substantive policy challenges.

Instead, the purpose has been to explore the dilemmas faced by the EU in relation to environmental policy. By the same token, it is recognised that actions within the existing framework will inevitably produce imperfect results and that practical alternatives are difficult to conceive. For some, the obvious remedy would be to reject the orthodoxy of consumer capitalism and to develop the debate on practical alternatives (Trainer, 1985; Daly, 1992; Seager, 1993; Paehlke, 1995). However, O'Riordan and Voisey's (1998) concept of the sustainability transition and Turner's (1993) spectrum of sustainability seem more obviously realistic frameworks to work within. As the European Union must protect all its legitimate interests and duties - the defence of the environment, the well-being of its citizens and the sovereignty of its

Member States - it is unreasonable to expect ambitious policy shifts. However, it is also important to recognise that the outcome of the present approach is likely to be weak sustainability. New policy instruments are unlikely to change this situation because they are essentially technical solutions when the most severe challenges concern how society perceives the importance of the environment.

Aside from the problem of values, it has been argued that neither the divergence in Member State policies nor the uneven standards of implementation have been remedied by the move towards economic instruments. The main force behind policy convergence has consistently been the imposition of legislative standards. Economic instruments have principally been a technical instrument for providing environmental expenditure revenue. More encouragingly, there have been only sporadic instances where EU institutional procedures have fundamentally hindered the effectiveness of economic instruments. The German and Danish cases show that the free-trade provisions of the EU Treaty *can* conflict with its environmental remit (which theoretically supports the waste management hierarchy) but, generally, economic instruments have been successfully assimilated within the existing structure.

Instead, the main factor determining the environmental efficacy of economic instruments has been the implementation methodologies employed by individual Member States. If, in fact, economic instruments do encourage the fragmentation of national standards, upholding the aims of the EU environmental programme will depend upon the legislation used to underpin them and the Commission's ability to enforce EU law. Both the Packaging Directive and the EU energy/carbon tax case suggest that economic instruments applied at a national level send uneven price signals to market systems. Therefore, without greater analysis by the Commission of the incentives contained in national taxes, the state-led approach may lead to further dislocations in the environmental *acquis*. There is some indication that fines for Member States could help to combat the implementation deficit but it will be some time before it is possible to determine whether they significantly reduce the number of infringements of EU environmental law.

Ultimately, the EU environmental programme is still evolving at a rapid rate. The new competencies and procedures bestowed in the Maastricht and Amsterdam Treaties have yet to take full effect. What is apparent, however, is that economic instruments can

serve a useful purpose within the wider programme, but they are not a panacea. Too little is known about their practical impact, making further research in this area a high priority. However, considering that the EU possesses neither the democratic legitimacy nor the institutional power to adopt more radical or centralised environmental policies, the most appropriate focus for the present must be on ensuring that existing measures are properly applied. This will provide a more solid basis for assessing whether the EU's existing policies and philosophy towards the environment are a sufficient response to the challenges of sustainable development.

Chapter Nine

Conclusions and Prospects for the Future

9.1 Introduction

The aim of this thesis has been to examine the processes shaping the implementation of European Union environmental policy. In particular, it has sought to present evidence evaluating the role of price-based regulation and the contribution of the EU's approach to environmental policy to the promotion of sustainable development. During the study a number of areas have been identified where economic instruments have led to major improvements in environmental standards. However, it has been argued that in key areas they have fallen short of their main objectives and the principles of sustainability. These failures can be attributed to a combination of factors; the tactical responses of businesses to environmental taxes, the assumptions made by policy-makers when designing price-based instruments, and the complexities of the EU's policy-making and implementation structures. To conclude this discussion and synthesise its findings, this chapter has three main objectives. The first is to review the study's key findings and success in achieving its aims. The second is to consider the future prospects for price-based environmental regulation in the EU. The final aim is to suggest further research opportunities that have emerged as a result of this investigation.

9.2 Main Findings and Evaluation of the Thesis

Within the broad remit outlined above, three main objectives were set for the research:

- To assess, in the case of the Packaging Directive, the efficacy of the EU's state-led style of policy negotiation and implementation in relation to the promotion of sustainable development;
- To investigate the extent to which price-based policy instruments have encouraged sustainable business practices;
- To assess the compatibility of the EU's political and decision-making structures with the effective operation of price-based environmental regulation.

9.2.1 EU Policy Implementation and Sustainable Development

There is broad agreement within the literature that the formulation of EU environmental policy fluctuates between a form of inter-governmental, lowest-common-denominator bargaining and a more confederal, entrepreneurial style of policy-making (Weale, 1999; Haas, 1998; Zito, 1998). There is also consensus that although ultimate control over the adoption of policies is vested in the Member States, no single institution or coalition consistently dominates the policy-making process. Instead, influence is sufficiently well distributed between the EU's major institutions that agreement must be reached between a wide range of actors before policies can gain acceptance (Weale, 1996). The Packaging Directive has proved no exception to these rules. The Commission's original proposals received the support of several environmental 'leader' states but were opposed by a more sceptical coalition headed by Britain and France (Golub, 1996). Because the latter group prevented the formation of a qualified majority in Council, several of the Environment Directorate's more ambitious proposals, including those for mandatory packaging reduction and re-use targets, were removed entirely, whilst its suggested recovery and recycling targets were significantly eroded (Bailey, 1999a).

The analysis therefore revealed a number of interesting points. The first is that whilst this division of responsibilities is a necessary part of democratic politics in the trans-national EU polity, it encourages a bargaining mentality amongst its key actors rather than one geared towards the 'rational' identification and resolution of environmental problems (Weale, 1996). This situation is inevitably fuelled by the inability of science to provide firm guidance on environmental problems, a knowledge vacuum which is often filled by interest-led arguments (Zito, 2000). The second point concerns conflicts between EU policies, particularly those relating to its environmental ambitions and trade commitments. In spite of the EU's general support for the waste management hierarchy, the Council was unable to agree waste reduction and re-use standards - two of the hierarchy's key elements - in the Packaging Directive. Mandatory reduction targets were considered to be economically damaging and unrealistic to implement, while re-use was deemed to restrict EU free trade. Notwithstanding the obvious practical difficulties of implementing meaningful reduction and re-use targets, it seems that the sustainability of EU waste policy was ideologically impeded by the Union's commitment to other policy priorities (Bailey, 1999a). Whilst acknowledging that all political agreements involve negotiation and compromise, it was argued that the

prominence of this bargaining mentality within the EU political system often counteracts the benefits gained from collective environmental decision-making.

However, the study's main finding in this area concerns the extent to which state authorities adapt EU legislation during the process of practical implementation. Although flexible implementation has been a constant and deliberate feature of EU environmental policy (Lowe and Ward, 1998b; Barnes and Barnes, 1999), it was argued that the way national authorities implement EU legislation has a major influence on their contribution to sustainable development. For example, the German packaging waste policy sought to promote stronger sustainability by adopting stringent legislation, centralised management and punitive environmental charges, though whether these objectives have been achieved is still the subject of heated debate (Staudt, 1997; Flanderka, 1998). The UK's consultative, cost-conscious and market-led approach has led to economic efficiency but weak sustainability outcomes. It therefore seems that assessing national policies purely in terms of legislative standards does not provide an adequate evaluation of their overall contribution to sustainable development. A key focus for future EU policy, therefore, needs to be the implementation methodologies employed by national and regional authorities (Jordan, 1999).

Ultimately, however, the conceptual ambiguity of sustainable development made it difficult to provide a categorical evaluation of the procedures used to implement EU policies (see Chapter two). Equally, the criteria used to evaluate sustainable development inevitably depend on the standpoint of the observer. In this case, the study was conducted from an environmentalist perspective (Chapter four). Whilst EU policy was assessed using criteria reflecting the diversity of the sustainable development debate, including Gibbs *et al's* (1998) definitions of weak and strong sustainability and O'Riordan and Voisey's (1998) concept of the sustainability transition, inevitably its findings - as with all social research - were influenced by the researcher's personal perspective (Kitchen and Tait, 2000; Williams, 2000).

9.2.2 Price-based Regulation and Business Behaviour

According to the existing literature, price-based environmental regulation serves two main purposes. First, environmental tax revenue may be earmarked for environmental, economic or social expenditures. Second, environmental charges can be used to

encourage industry or consumers to reduce their environmental impact (Baumol and Oates, 1988; Gee and von Weizsäcker, 1994; van den Bergh, 1996). The study showed that while the German and British schemes have generally been successful in generating hypothecation revenue, both have experienced problems in allocating resources to priority areas. The main difficulty in Germany was that insufficient attention was paid to the development of reprocessing infrastructure. In Britain, the problem has been in ensuring that private-sector reprocessors use PRN revenue for environmentally related purposes (Bailey, 1999b). Both governments have sought to resolve these issues by increasing the regulation of their packaging waste management systems. Whilst the centralised structure of the Dual System made it relatively straightforward for the German government to adjust both its regulatory controls *and* Green Dot prices, the situation in Britain has been complicated by the government's (and industry's) preference for market-based pricing.

The conclusion, therefore, is that since the source of many environmental problems lies in the failure of market systems to acknowledge the value of environmental resources, the utmost care must be taken when introducing market-based solutions. Because 'free' markets are founded on profit maximisation rather than social justice or environmental sustainability, government intervention must ensure that such factors are not neglected. Conversely, Britain's market-led scheme has proved relatively economically efficient while Germany's 'command-and-control' regime has placed a major economic burden on its industries and possibly reduced their capacity and willingness to accommodate environmental regulation in other areas (Knabe, 1995).

The survey of packaging producers (Chapter six) demonstrated that German businesses are more heavily engaged than their British counterparts in each element of waste management promoted by the Directive. Two aspects were particularly significant in terms of sustainable development; the proportion of businesses involved in packaging waste reduction (57.1% in Germany, 12.7% in Britain) and the development of end-use markets for recycle (53.2% of German companies, 24.5% of British firms). However, in terms of the incentive effect created by environmental taxes, no significant correlation was found in either country between packaging waste charges and business willingness to participate in activities which reduce landfill disposal. Two primary explanations for this were proposed. First, it was suggested that packaging charges were too low in relation to the logistics and marketing benefits of packaging to have a

major bearing on business decisions (Jones, 1999). Second, many businesses, particularly within Germany, were recouping at least a proportion of recycling costs through increased product prices and, by doing so, were disseminating but diluting the incentive effect of environmental charges. Therefore, contrary to the claims in the economics literature, the study concluded that economic instruments produce only marginal changes in the behaviour of industry and that those observed were mainly produced by legislative requirements and the coercive power held by national authorities. This is not to say that economic instruments cannot help in the fight against pollution, but that their capabilities should not be over-estimated. As More *et al.* (1996: 407) comment:

Decisions can be made on various grounds: economic, moral, aesthetic or rational. In our society, however, economics has become the language of default. We are unused to dealing with other grounds for decisions so, when confronted with a difficult choice, we turn quite naturally to economics - surely one of the most sophisticated and powerful tools for decision-making ever devised. Yet this may lead us to extend economics to areas where it is ill-suited to serve, a disservice both to the issue at hand and to economics.

However, as well as the importance of these conclusions, it is also important to recognise the study's limitations. The first caveat is that the analysis examined a single piece of legislation. Some of the problems highlighted, such as the recovery of used materials through the supply chain and the development of end-use markets, clearly do not apply outside the waste management industry. Similarly, the research investigated one variant of price-based regulation. Its findings may therefore not be valid for tradeable permit systems, subsidies or other economic instruments. However, it would be premature to dismiss this as an idiographic study. Firstly, packaging waste is only one element of a wider waste management problem. Several existing waste taxes - such as the UK landfill tax and the Danish trials with weighed refuse collection¹ - also seek to change industry or consumer behaviour. (Powell and Craighill, 1997). Moreover, PRN-style mechanisms are being considered for other waste streams, including end-of-life vehicles, waste electrical and electronic equipment, batteries and biodegradable waste (DTI/DoE, 1991; 1992; DoE, 1993; Pearce and Turner, 1993;

¹ Under these trials, weight-based charges were introduced for non-compostable household waste. At the start of the scheme, waste was weighed on collection and households were charged at the same rate as the waste collection element of municipal taxes. The result of the direct charge has been a 15-20% reduction in the weight of non-compostable refuse collected. As the charge rate did not change, the reduction is attributed to households taking the opportunity to reduce their bills by separating compostable materials and increasing their participation in recycling (see Barrett *et al.*, (1997:112)).

Fenton and Hanley, 1995; Palmer *et al.*, 1997; Turner *et al.*, 1998; DSD, 1999c; MRW, 1999f; Thurgood, 2000). That said, more research is undeniably needed in order to explore the issues raised by this study further.

The study's methodological limitations should also be acknowledged. The most obvious concern was whether the sampling frames used in the producer survey were representative of the overall population of businesses affected by packaging regulation. Respondent self-selection is always a problem; for example, did only the most proactive German companies return the questionnaire and how far did non-responses influence the results? Similarly, some doubts were expressed about the comparability of the two respondent groups. Whilst Chapter six showed both samples to be generally representative of the sectors targeted by packaging regulation, it proved difficult to compare the two groups in areas where the two sets of legislation diverged. Similarly, there is the problem of respondent validation in postal surveys. Though every effort was made to minimise the need for respondent interpretation, it is always possible that questions may have been misconstrued or misunderstood.

In terms of the evaluation methods used, some economists consider the assessment of business responses to environmental taxes in terms of turnover as controversial or simplistic. Whilst it is a relatively crude measure and would benefit from further refinement, these objections seem slightly conjectural considering the number of businesses that evaluate their environmental strategies in precisely this manner (Watkins, 2000). Finally, although this study's remit was to examine the influence of environmental taxes on business practices, there were clearly other important variables, such as the stringency of legislative standards, the length of time legislation has been in place, and differences in corporate social-responsibility culture. Whilst these factors were considered briefly in Chapter seven, it would be useful for future studies to develop broader evaluation frameworks.

9.2.3 EU Institutional Procedures and Economic Instruments

The third objective of the study was to assess the compatibility of the EU's decision-making structures with the effective operation of price-based environmental regulation. The main finding was that there have been few conflicts between nationally-implemented charges and other aspects of the EU Treaty in the case of the Packaging

Directive. Two reasons for this were suggested; first, the EU institutions have no power to dictate the fiscal measures used by Member States to implement EU law and, second, most national packaging taxes are, by design or default, compliant with EU free-trade rules. Clashes between the Commission and Member States have only occurred where the Commission believes national legislation contravenes Articles 94-5 of the Amsterdam Treaty. Even the most controversial of these, the deposit-return scheme for beverage containers in Germany, is not opposed on principle but because it imposes disproportionate transport costs on non-German manufacturers.

However, if the trend towards price-based regulation continues, there is a danger that it may remove some momentum from the push-pull dynamic of EU policy-making. This dynamic begins where environmental 'leader' states introduce new standards then, in order to protect national competitiveness, seek to have them adopted across the EU. Such proposals are often resisted by 'laggard' states on the grounds that they are either environmentally or economically unjustified. The result, usually, is the adoption of a diluted proposal and an incremental increase in environmental standards across the Union (Sbragia, 1996). If the free-trade argument becomes less relevant under price-based regulation, the pressure for harmonisation may also diminish. However, it is unlikely that the pressures creating the push-pull dynamic will evaporate entirely, not least because most economic instruments are introduced as part of wider EU commitments under the EAPs and international agreements; hence, all parties are working within common frameworks. Therefore, whilst there appear to be few conflicts between price-based environmental regulation and the EU Treaty, economic instruments may further problematise the already cumbersome processes of policy formulation and implementation.

Chapter eight then examined the paradoxes inherent in the EU's attempts to fit its sustainability ambitions alongside its longstanding commitments to economic and trade development. As a liberal-democratic system founded on capitalist ideals - and moreover, according to W. Wallace (1996) and Höreth (1999), as an incomplete political entity - the EU cannot legitimately impose the kinds of restrictions on welfare and freedom that may be required to achieve sustainable development (Jacobs; 1994, Musschenga, 1994). It was argued that even assuming agreement could be reached on the nature and seriousness of environmental problems, EU decision-making processes are ill-equipped to agree and justify the necessary courses of action, especially where

these might entail radical changes to society's values and objectives. As price-based regulation is currently applied within the wider neoliberal-capitalist paradigm of economic expansion, the technique is likely to ameliorate environmental problems rather than solve them. However, whatever the limitations of the present system, it was acknowledged, first, that the EU programme has produced major advances in environmental protection and, second, that there are few immediately practical or morally defensible alternatives to liberal democracy. Therefore, notwithstanding the conflicts within EU policy, the chapter concluded that the immediate priorities should be to ensure that existing measures are fully and faithfully applied and to assess their environmental efficacy against the principles of sustainable development (Barnes and Barnes, 1999).

One final task in appraising the work of this thesis is to highlight its contribution to contemporary debates in human geography. It is widely acknowledged that sustainable development embraces a broad range of issues; indeed, its defining feature is its attempt to construct a holistic analysis of interactions between economy, society and environment (Bell and Morse, 1999). It is therefore apparent that interfaces between geography, economics, management studies, policy studies and the natural sciences are an integral part of exploring sustainability issues. This thesis has demonstrated particularly interesting links between economic, industrial and environmental geography by examining the structural and practical issues surrounding the environmental sustainability of economic and political processes. It has sought to illuminate the ways in which environmental, political and economic issues become entwined in the EU and the effects of these on the practical application of environmental policies. Furthermore, as these interactions occur at several levels in EU, national, regional and business planning, each stage of policy implementation presents its particular challenges and conflicts of interests and priorities.

Another important contribution comes in terms of the application of regulation theory to environmental problems. The regulation theory approach focuses on relationships between the processes of accumulation that characterise modern capitalism and the ensemble of institutional processes that comprise the mode of social regulation (Peck and Miyamachi, 1994; Gibbs, 1996). More specifically, it explores the manner in which social regulation guides and stabilises accumulation processes so as temporarily to avoid capitalism's crisis tendencies produced by its natural disregard for externality

effects (Baldwin and Cave, 1999). Gibbs (1996) suggests that regulation theory has particular application in respect of sustainable development because of the latter's specific integration of environmental, social and economic concerns into a holistic analytical framework.

Of course, regulation theory is not solely a geographical construct. Baldwin and Cave (1999) note that it has been employed in such diverse disciplines as law, economics, political science, history, psychology, management and social administration. However, a regulation theory approach to geographical analysis of environmental policy is especially useful in understanding the scale and limitations of institutional regulation with respect to environmental externalities. In terms of scale, it appears that strategic environmental regulation is shifting towards the EU, a move which is a corollary of the region's economic integration. Within this context, more concrete and tactical forms of regulation have been applied by the Member States. However, the important point is that the framework of regulation and broad ideological conflicts between competing capitalist and environmental interests are increasingly being contested at a supranational level. If one accepts the notion that the EU is primarily organised around economic imperatives, this suggests that the current organisation of regulation may tend towards weak forms of sustainability unless it is also combined with changes in the values and attitudes of key decision-makers (Goodwin et al., 1995).

It is also important to note the limits to effective social regulation during the practical implementation of environmental policies. The central question here concerns the extent to which industries operate within, or autonomously of, national, regional, or even international institutional checks. There is already a lengthy literature documenting the loss of national control over economic activity resulting from globalisation (for example, Gwynne, 1985). The evidence from this investigation suggests that the immense resources required in the modern state to monitor the environmental behaviour of businesses at the local level significantly impairs the state's effective regulatory capacity. This problem of scale - which was evident in both Britain and Germany - coupled with the UK government's desire to seek quasi-market solutions to environmental problems, indicates that significant transformations in values and attitudes will be necessary in many areas before the tensions between economy and environment are satisfactorily eased. Approaching regulation theory from a geographical perspective has therefore aided the exploration of many of the

structural and spatial complexities of implementing sustainable environmental and economic policies.

Again it is dangerous to make broad generalisations about the nature of EU environmental governance from a single study. Further research on the relationship between price-based regulation and EU policy processes is required before this is possible. This research has therefore, as much as any individual work can, made good progress against its main objectives and raised important questions about the practicalities of environmental policy implementation. However, as Chapter eight noted, there are few answers that can neatly unravel the relationship between environmental and economic policy or their management within the complex entity of the European Union. As O'Riordan and Voisey (1998: 3) eloquently put it:

The sustainability transition ... is the process of coming to terms with sustainability in all its deeply rich ecological, social, ethical and economic dimensions. The transition is as much about new ways of knowing ... as it is about management and innovation of procedures and products. As a species, we have barely begun to imagine how to think sustainably.

In exploring some of broader issues surrounding the practical attainment of environmental sustainability, as well as through its empirical work, this study has sought to advance this process of debate and learning.

9.3 Prospects for the Future

Having reviewed the study's key findings and their research contribution, the chapter now considers the outlook for price-based regulation in the EU. Two issues are likely to assume particular importance in the coming years. The first concerns the flexibility of the EU environmental programme and, more expressly, the prospects for co-ordinating Member-State economic instruments. The second acknowledges the changing composition of the EU and the challenges posed by the accession of the Central and Eastern European (CEE) states. The following section examines how each of these dynamics is likely to affect the political balance of environmental policy in the European Union.

9.3.1 Flexibility and the Implementation of EU Environmental Policy

This thesis has highlighted a number of difficulties stemming from the EU's flexible, state-led style of environmental policy. However, it would be wrong to claim either that tangible controls do not exist (the EU Treaty, the EAPs and EU law are all significant and binding commitments) or that flexibility has not yielded major benefits. It has been instrumental in maintaining a balance between inter-governmentalism and confederalism within the EU and, moreover, it has made environmental policy sensitive to the wishes and capabilities of its constituent powers (Jordan, 1999). More worryingly, state-defined implementation has led to uneven and, arguably, diverging environmental standards in the Member States. For reasons explained previously, this is a trend which economic instruments seem unlikely to reverse. These concerns are further compounded by the increasing number of infringements of EU environmental law in recent years (Demmke, 1997; Haas, 1998; Barnes and Barnes, 1999)². Therefore, although there are sound reasons for supporting the EU's flexible approach to environmental policy, the fear must be that the programme in its current format will struggle to maintain its coherence and focus on sustainable development.

Whilst implementation flexibility is a necessary and integral part of environmental policy management in trans-national institutions, this does not preclude improvements being made to the present arrangements. Considering the record of poor implementation, tackling monitoring and enforcement is an obvious priority. However, as this issue falls largely outside the scope of this thesis (see House of Lords (1997) for a fuller discussion of the role of national authorities and the EU in environmental policy enforcement), this section focuses on the co-ordination of price-based regulation in the Member States.

The first issues to consider are what options might be available and what modes of control might be politically, economically and environmentally acceptable to all Member States. Obviously there is no constitutional mandate in the Treaty enabling the EU institutions to prescribe how Member States should implement environmental law. Nor would such a 'top-down' approach be effective. It would be more likely to

² According to the latest Commission survey, 29% of all suspected infringements of EU law and 25% of open court cases in 1999 related to environmental policy. Furthermore, 45% of these concerned the non-respect of ECJ judgements. It would appear that the problem is mainly one of poor implementation, as the transposition rate of EU environmental law was 97% compared with an average of 94.5% over the twenty policy areas surveyed (ENDS, 2000d).

be insensitive to local circumstances and to make policy implementation expensive, especially where national authorities were forced to adopt unfamiliar or inappropriate administrative procedures (Jordan, 1999). Similar objections prevent the Commission taking a doctrinaire stance on the design and setting of environmental taxes. Even if the constitutional difficulties were somehow waived, the centralisation of environmental taxes would require unanimous Council approval under Article 93 of the Amsterdam Treaty. Decision-making in this area would therefore effectively be paralysed. Furthermore, since the costs of pollution reduction vary between individual polluters and countries, uniform charges (and standards) would have a disproportionate effect on less-developed Member States (Manser, 1993). On this basis and the premise that nationally-led economic instruments cause few problems for inter-state trade, the logical approach is for the Commission to encourage differentiated environmental charges where they promote an appropriate balance between environmental protection and economic development in each Member State (Halkos, 1996).

If it makes little economic or environmental sense to adopt a overbearing approach to policy implementation, future co-ordination must be limited to the definition of general rules for the application of environmental taxes. Whether or not a less inter-governmental outlook eventually emerges hinges largely on the extent to which Economic and Monetary Union (EMU) encourages the federalisation of fiscal policies. This year, EU finance ministers 'broadly endorsed' new criteria for the introduction of tax cuts in the Member States, the general aim of which is to maintain budgetary discipline within the Single Market (Financial Times, 2000: 1). Although some observers fear this may undermine national sovereignty on tax issues, it demonstrates that greater fiscal co-operation is possible. An indication of this trend in relation to environmental policy came in the Commission's reports to the 2000 Inter-Governmental Conference (IGC); this recommended the extension of QMV to more areas of environmental policy, including taxes raised purely for environmental purposes (IGC, 2000a; 2000b). Under this proposal, QMV could be expanded to all environmental polices except those related to town and country planning and land use (aside from waste policy).

If such streamlining of EU environmental decision-making comes to fruition, there are a number of areas where national environmental taxes might co-ordinated. First, where EU legislation advocates the adoption of economic instruments, the creation of banded

environmental taxes could be considered. The aim here would be to make tax systems responsive to the needs of individual states whilst ensuring they are based on scientific evaluations of environmental and economic impact rather than short-term political objectives. If this were still considered too prescriptive, another option would be to require Member States to demonstrate, in advance of implementation, the link between their environmental taxes and the sustainability outcomes sought by EU legislation. To do this in retrospect, as has occurred with the Packaging Directive, only adds to the time between the initiation of legislation and the first evaluation of its effectiveness³. Whilst Jacobs (1991) notes that the effect of environmental taxes can rarely be guaranteed - making charge iterations inevitable - excessive reliance on a dose-response methodology can be expensive for industry and deter investment in pollution abatement (as Britain's experience with PRNs testifies, also Welford and Prescott, 1994). It is therefore seems logical that the environmental and economic effects of economic instruments be more fully evaluated before they are applied in practice.

The second option would be to introduce rules governing the incentive patterns created by environmental taxes, a key element of which should be the strengthening of rewards for improved environmental performance. This objective might be pursued either through increased environmental taxes (the German approach in the Packaging Ordinance) or the greater use of subsidies (Jacobs, 1991). Subsidies are generally employed as financial rewards for reduced pollution, or as grants or 'soft' loans to businesses investing in environmental protection (Rees, 1997). Though both the British and German packaging waste models grant *de facto* subsidies to their reprocessing sectors, this study has shown that recycling charges have produced only marginal changes in polluter behaviour. The question is therefore whether an alternative tack, the use of subsidies to recycle taxes back to producers, would stimulate a greater response. This approach is similar to that employed, admittedly with variable success, by the Environmental Bodies Credit Scheme under the Landfill Tax (HCSCETRA, 1999). The subsidies approach does create its own problems, however, as subsidies would either act as a drain on public finances or restrict the revenue available for other environmental investment. Furthermore, the misuse of Landfill Tax credits by some landfill operators serves notice of the dangers of under-regulated subsidy schemes

³ In 1998, the Environment Directorate of the European Commission instituted a project to evaluate the implementation of the Packaging Directive in the fifteen Member States (E3/ETU/980111). The results of this are scheduled for publication in late 2000, four years after the transposition deadline in the Member States.

(ENDS, 1998g). Nonetheless, by providing positive incentives to industry, subsidies may facilitate their engagement in pollution prevention.

The third option would be to introduce general rules on the hypothecation of environmental taxes. The aim here would be to prevent recurrences of the PRN situation, where individual market players were able to exploit regulatory loop-holes and manipulate the hypothecation system (see Chapter five). Learning from the British experience, the Commission could require that Member States meet two pre-conditions. First, they should ensure that recipients of hypothecated revenue account for the proportion directed towards environmental expenditures (DETR, 1998a; 1999a). Second, national authorities should specify the uses revenue may be put to under the hypothecation scheme. This, it has been argued in Chapter seven, should include an element of pollution prevention. A more ambitious tactic would be to specify that minimum percentages of tax revenue should be diverted towards environmental projects, though this is again likely to be too prescriptive and susceptible to manipulation.

In the final analysis, it is doubtful whether the Member States are ready to accept greater EU involvement in the practical implementation of environmental policies as national sovereignty on fiscal matters and the subsidiarity principle remain politically sensitive issues. The difficulty, as Höreth (1999) sees it, is that much of the 'zero-sum' political discourse considers that strengthening one institution (the EU decision-making process) by definition weakens the other (national processes) (also Zito, 2000). For this reason, only general suggestions for policy co-ordination have been made here. However, the danger is that a disjointed EU environmental programme will struggle to retain its coherence and, more specifically, its focus on sustainable development. This suggests that a degree of mutually agreed co-ordination (in addition to that already imposed by legislative and Treaty requirements) should be extended to policy implementation. In order to make this process consensual rather than authoritarian, national authorities should make the agreement of common implementation rules a key component of the policy-negotiation process. Whilst this will undoubtedly complicate policy-making rules further, it should help to reduce disparities and produce a more 'level playing field' in relation to environmental policy and EU free trade.

9.3.2 *Enlargement*

Whilst this thesis has concentrated mainly on the EU's internal political dynamics and their influence on environmental policy, the changing geopolitical structure of Europe since the late 1980s has brought new dimensions to the environmental debate (Barnes and Barnes, 1999). The most influential of these is likely to be the accession of new Member States to the Union (Blacksell, 1998). This is especially true considering that, of the thirteen states currently applying for EU membership, the majority are from Central and Eastern Europe (CEE) (CEC, 1997d)⁴. Many of these countries suffered appalling environmental mismanagement during the Soviet era and are undergoing slow and painful transitions to Western political and economic philosophies (Klarer and Francis, 1997; Blacksell, 1998; Saiko, 1998). Two issues concerning the CEE accessions are of particular importance in the context of this thesis. The first is how enlargement will affect the EU's already complicated decision-making procedures. The second concerns the role of price-based regulation in the environmental rehabilitation of the CEE states.

The first stage in the accession process is for CEE states to harmonise their laws with those of the EU. The view of the European Commission, expressed at the Environment Council in 1996, was that this should include all aspects of the environmental *acquis* (Mayhew, 1998). This process is expected to take at least ten years in some cases because of the sheer scale of the work and the costs involved (Turner, 1997, cited in Barnes and Barnes, 1999; Saiko, 1998). According to the Commission, even Slovenia, which enjoys one of the most prosperous economies in Eastern Europe, faces immense practical and institutional challenges in conforming to EU environmental standards (CEC, 1999c). The strategy proposed at the EU's Cardiff Summit in 1998 was that some of the funding for environmental approximation in the CEE countries should come from the PHARE programme⁵ but that the majority should be financed by the applicant states themselves (Barnes and Barnes, 1999). However, a number of other bodies have also established funds to aid environmental and economic transition in the

⁴ According to the current timetable, the first wave of accession will include the Czech Republic, Estonia, Hungary, Poland and Slovenia. The other states being considered for EU membership are Bulgaria, Cyprus, Latvia, Lithuania, Malta, Romania, the Slovak Republic and Turkey (Barnes and Barnes, 1999).

⁵ The PHARE programme was set up to support economic and political transition in Poland and Hungary. In 1999, it had a budget of 6.693 billion Euros (<http://europa.eu.int/comm/enlargement.htm>).

CEE nations, including the European Investment Bank and the Global Environmental Facility (Klarer and Francis, 1997).

Aside from the practical challenges posed by enlargement, each wave of accessions alters the balance of power in the EU institutions and, potentially, the direction of the environmental programme (Barnes and Barnes, 1999). The last enlargement in 1995, which incorporated Austria, Finland and Sweden, might reasonably be argued to have increased the influence of the 'green' lobby, though Liefferink and Andersen (1998) claim that consistent alliances between environmental-activist states have failed to materialise because of differences in strategic outlook. However, the fact remains that it will be more difficult to reach agreements in an EU comprising twenty or even thirty members. Although one should not automatically assume that the CEE governments will be a negative influence on environmental policy, their presence will undoubtedly alter the balance in the Council and the European Parliament (Table 9.1) and overshadow the pro-environmental shift produced by the 1995 accessions (CEC, 2000b). Considering the transitions taking place in CEE states, the fear must be that they will seek to slow the pace of environmental policy in order to protect their fragile market economies. Under present voting rules, this could lead to policy stagnation and the dissipation of the push-pull dynamic. Less sensationally, lowest-common-denominator bargaining could re-emerge as the predominant form of policy-making.

Pre-empting such contingencies, the issue of expanded policy-making was one of the few environmental issues debated at the EU's 1996/7 Inter-Governmental Conference. The main theme here was the need for greater policy flexibility, first, to help manage the lengthy transitions required by some prospective members and, second, as a route forward for environmental policy where not all Member States wish to take part (Lowe and Ward, 1998b). Whilst the practical reasons for this tactic are plain, it opens up the prospect of a two or even three-tier EU environmental policy and could upset the delicate balance between Union's free-trade and environmental policies.

Table 9.1 The Provisional Division of voting rights under an expanded EU

<i>Member State</i>	<i>Population (millions)</i>	<i>Population (%)</i>	<i>European Parliament Seats</i>	<i>Council Votes</i>	<i>Commission</i>
<i>Existing</i>					
Belgium	10.21	1.87	25	5	1
Denmark	5.31	0.97	16	3	1
Germany	82.04	15.04	99	10	2
Greece	10.53	1.93	25	5	1
Spain	39.39	7.22	64	8	2
France	58.97	10.81	87	10	2
Ireland	3.74	0.69	15	3	1
Italy	57.61	10.56	87	10	2
Luxembourg	0.43	0.08	6	2	1
Netherlands	15.76	2.89	31	5	1
Austria	8.08	1.48	21	4	1
Portugal	9.98	1.83	25	5	1
Finland	5.16	0.95	16	3	1
Sweden	8.85	1.62	22	4	1
UK	59.25	10.86	87	10	2
TOTAL	375.36		626	87	20
<i>Applicants</i>					
Bulgaria	8.23	1.51	21	4	1
Cyprus	0.75	0.14	6	2	1
Estonia	1.45	0.27	7	3	1
Hungary	10.09	1.85	25	5	1
Latvia	2.44	0.45	10	3	1
Lithuania	3.70	0.68	15	3	1
Malta	0.38	0.07	6	2	1
Poland	38.67	7.09	64	8	2
Czech Republic	10.29	1.89	25	5	1
Romania	22.49	4.12	44	6	1
Slovakia	5.39	0.99	16	3	1
Slovenia	1.98	0.36	9	3	1
Turkey	64.39	11.80	89	10	2
TOTAL	545.56	100	963	144	35

Source: CEC (2000b: 63)

The second issue concerns the ability of the CEE countries to cope with the strictures of price-based environmental regulation. As part of the association agreements, the Commission routinely measures the progress of each applicant state against two economic criteria, the existence of a functioning market economy and their capacity to withstand competitive forces within the Union (CEC, 1999c). The Commission

currently considers that the Czech Republic, Estonia, Hungary, Poland and Slovenia have met the former requirement and that Slovakia is very close. Against the second standard, the Czech Republic and Slovenia have made greatest progress, Hungary and Poland are proceeding apace and Estonia still has considerable work to complete. The Commission's view is that other CEE states could catch up within ten years if determined efforts are made. At the same time, the report expresses concerns about corruption and breakdowns in the rule of law, particularly in Romania.

However, Britain's experiences with the PRN system demonstrate that the existence of a market economy alone is not sufficient to assure the success of price-based environmental regulation. For economic instruments to succeed, market systems must be able to allocate resources efficiently and in accordance with government policy objectives. The extent to which either precondition can yet be met by the CEE states is questionable, as legal and administrative weaknesses have created serious obstacles to the approximation process (Barnes and Barnes, 1999). The environmental impact of industrial activity is still severely under-regulated in most CEE countries, whilst privatisation and the transition to the market economy have squeezed profits and intensified competitive pressures (Saiko, 1998). Neither of these bode well for the formal internalisation of environmental costs, whilst endemic corruption in some national authorities makes the task of effective enforcement even more daunting (Saiko, 1998)⁶. Manser (1993: 93-4) suggests that in Poland, for example, establishing 'tough enforcement by a competent local ecological "police"' is a higher priority than any switch from legislative to price-based regulation. More optimistically, some reports claim that the expansion of trade between EU and CEE countries has forced many CEE manufacturers to adopt EU product standards even where the enabling legislation has yet to be transposed (EAP Task Force, 1995).

Nonetheless, aside from the immediate clear up, the most urgent priority is to develop institutions and procedures capable of effective environmental management, as only then will it be possible to support the wider use of market-based environmental

⁶ Aside from corruption, Manser (1993) discusses three main obstacles to the effective monitoring and enforcement of environmental policy in the CEE states. First, the decentralisation of monitoring activities compared with the main decision-making apparatus in most CEE countries means that local enforcement offices lack the necessary staff and skills to enforce legislation. Second, many industrial plants simply lack the necessary control equipment. Third, some CEE governments have allowed industry to side-step fines or charges in order to protect their country's economic performance in times of recession.

regulation (Klarer and Francis, 1997). Though many CEE governments have increasingly deployed PPP-based environmental regulation since 1989, including measures to regulate packaging waste in the Czech Republic, Estonia, Hungary, Latvia and Slovakia (Perchards, 1998), the correct sequencing of events is critical to the economic and environmental revival of Eastern Europe. Attempts to expand market-based regulation before legislative frameworks and enforcement competencies are in place could damage both causes (EAP Task Force, 1995). The pitfalls of 'shock therapy' have arguably been a key lesson from the transition of former socialist states to the market economy. Although it is essential that the value of environmental resources are not neglected in the CEE economies, it seems prudent that the former socialist states and those directing the accession process in the EU pursue environmental capacity-building, particularly in respect of enforcement, before embarking on an ambitious programme of market-led regulation.

9.4 Towards a Research Agenda

This thesis has explored a range of issues concerning the implementation of EU environmental policy. It is hoped that the ideas expressed have relevance beyond the academic community, and that they will prove useful to policy-makers and industry managers alike. Inevitably, considering the complexity of the issues discussed, it has been impossible to provide an exhaustive analysis of the subject. Like most studies, it has left questions unanswered and has proposed conclusions that require further investigation. It is therefore appropriate to conclude the thesis by outlining a suggested research agenda for the future.

Unquestionably the highest priority is for further research examining business responses to price-based environmental regulation. Although the study produced important conclusions concerning the incentive and hypothecation functions of environmental charges, further research would improve understanding of these complex relationships. The first suggestion, therefore, is that the empirical work begun in this study should continue in the form of further industry surveys. These should focus on comparing different types of economic instrument and regulatory regimes in order to test the wider applicability of the tentative conclusions proposed in this study. It would also be useful to gain a more in-depth appreciation of industry's opinions of price-based environmental regulation. By aiming for a broad picture using postal

surveys, the reasons why businesses react to regulatory pressures in the way they do, or what might persuade them to engage with the sustainability debate more seriously, could not be explored fully in this research. The second strand of the proposed research agenda, therefore, is the use of qualitative methodologies to examine business reactions to price-based regulation.

The general aim of employing qualitative research here would be to further our understanding of the processes influencing business environmental thinking and practice. For example, business sector, geographical location, corporate culture, supply-chain pressures, stakeholder groups both within and outside the organisation, and local or regional government policies are all potential influences on the environmental behaviour of firms. Each of these may act as barriers to, or opportunities for, more sustainable business practice. Attempts to explore such issues using quantitative methodologies and postal surveys would involve high levels of interpretation by research participants and would, as a likely consequence, yield incomplete and unreliable data.

Semi-structured interviews offer a clear way forward in this context. Here common themes could be examined whilst enabling important or unexpected issues to be broached and elaborated upon. When exploring why respondents, corporate or otherwise, think or behave in a particular way, adopting a more inductive approach is often less constrictive than structured quantitative questioning. Proposition sets may also be included within interviews to help maintain structure whilst not stifling open debate. While there are always problems in generalising interview data gained from comparatively few respondents, this can be partly overcome by focus groups. These would enable groups of business managers to interact in identifying and prioritising potential barriers to sustainability. Finally, case studies could be used to examine business responses to different forms of environmental regulation (this technique was employed during the Masters pilot study). The main advantages of this technique were, first, that it provided the opportunity to talk to various members of the organisation and gain a broader view; second, that it helped to overcome the time constraints often imposed at interviews; and, third, that it provided first-hand experience of the difficulties companies face in complying with environmental legislation. Participant case studies were therefore an extremely effective way of exploring issues in greater depth.

There are also several potentially useful research avenues concerning the formulation and implementation of EU environmental policies. Perhaps the highest priority here is for research examining the CEE accessions and their impact on EU policy formation. Whilst this chapter has proposed that EU expansion may cause serious problems for the environmental programme, it will be some time before this can be fully assessed. Barnes and Barnes (1999) argue that even the influence of Austria, Finland and Sweden on environmental policy has yet to be properly determined. Two research possibilities can therefore be suggested. The first would be to examine the effect of the CEE accessions on the institutional dynamics of the EU and, more specifically, on the push-pull dynamic, subsidiarity, consensual bargaining and decision-making by concurrent majorities. The second would be to explore the implementation capabilities of the CEE states, where particular attention needs to be paid to the application of price-based regulation and enforcement issues.

A more local-level research opportunity not pursued in this thesis but alluded to in Appendix Seven is further exploration of environmental policy implementation at the intra-state scale. Appendix Seven, by noting the distribution of reprocessing facilities in Britain and Germany, demonstrated some clear patterns. For materials such as aluminium and steel, for example, only a few accredited reprocessing centres existed. For others, such as paper and plastics, there was distinct clustering around major urban centres with relatively few found in rural and peripheral areas.

It is immediately obvious that such patterns are in response to clear economic drivers. For example, the distribution of steel and aluminium reprocessors undoubtedly reflects economy of scale factors. Large volumes of materials are required to make such recycling operations viable and, in fact, most are integrated production and reprocessing facilities. The location of centres for other materials is also in response to the need to be close to source markets, in this case waste materials collected from industrial and household sources.

However, whilst such processes are fairly straightforward to understand, their implications for 'sustainable' waste policy and management require further investigation. The first issue to consider is the application of the proximity principle in waste management policy (Porter, 1998). The key questions here are, first, whether the

current distribution of reprocessing facilities is an appropriate interpretation of the proximity principle in terms of reducing environmental impacts from transport and, second, whether the policy instruments in operation encourage reprocessing or disposal of waste as close as possible to its point of origin. From the evidence in Chapter five, it would appear that the PRN system is less locally focused than the Dual System as it does not oblige waste producers to use the nearest facility if it is not the most economic.

However, any detailed examination of the proximity principle at an intra-state level would not be straightforward. The first task would be to produce inventories of waste movements from producer to reprocessor in order to ascertain the distances wastes are transported. Such audit trails are highly complex and information may often be incomplete, especially where waste producers or their agents are not required to use the nearest facility. Commercial confidentiality is also an obvious concern. It is therefore suggested that only relatively local-scale studies are likely to produce reliable results. Moreover, evaluating whether particular networks conform to the proximity principle will require assessment in the context of Best Practicable Environmental Option, balancing a range of economic and environmental factors, rather than solely against proximity criteria. Careful research scoping and design will therefore be of paramount importance in such studies.

The second issue is how the spatial concentration of reprocessing affects waste management and sustainability in rural and other peripheral areas. One topic of particular significance is the development of recycling in the former East Germany, where the data suggest that relatively few reprocessing centres are located in the new Länder. This apparent imbalance raises a number of questions relevant to both environmental and economic geography. For instance, does waste transportation to the West create a significant environmental impact? Has this imbalance led to lower recycling in the East and increased landfilling, fly-tipping and other environmental problems? Finally, to what extent have the new Länder lost out on sustainable waste management as a major growth industry in Germany? Similar issues could also be examined in Britain, particularly in areas such as the South West, which has few reprocessing facilities and its own well-documented economic problems.

Again, however, a note of caution is necessary. As well as the complexities of data collection already identified, it is clear that waste management is taking on increasingly international dimensions. This was demonstrated by the use of foreign reprocessing centres by UK reprocessors to obtain recycling certificates. This means that analysing waste management strategies purely on an intra-state level has its limitations without analysis of the wider national and international contexts. Even so, there remain numerous opportunities to investigate waste management policies and processes from both a bottom-up and a top-down perspective.

The final area of potential future research concerns recent developments in EU environmental policy-making. Many issues have been discussed in this investigation without being fully resolved. For example, to what extent will extended QMV streamline environmental policy-making? How much will framework directives and the Inter-Governmental Conference's resolve to make environmental policy more flexible affect the Member-State environmental standards? If there is greater co-operation in the design and implementation of environmental taxes, how will this process be managed and what effect will it have on progress towards sustainable development? To what extent can or should the implementation methodologies employed by the Member States be co-ordinated at EU level? Finally, how might the EU realign its fundamental values to better reflect those of sustainable development? It is therefore clear that this research is presented as much in the spirit of a beginning as of an end. Some issues have been clarified, new ones have emerged and much remains to be done. If these opportunities are taken up by other researchers, the coming years will witness an important advance in the study of environmental policy implementation.

It is also important to stress that human geography can make a substantial contribution to this research agenda. Its most obvious role is analysis of the spatial patterns caused by different environmental policies. This is particularly important considering the uneven patterns of environmental protection in the EU Member States and the further disparities that are likely to result from the CEE accessions. Gibbs and Healey (1997: 199) suggest that spatial analysis of environmental issues should take at least two forms. First, it should 'consider how incorporating sustainable development in both weak and strong forms impacts upon economic development over space' and, second, it should provide guidance on the most appropriate spatial scale to implement policies

aiming towards sustainable development. Finally, it has been argued that while sophisticated techniques have been developed for *predicting* how policy instruments *should* operate, many of these predictions have never been rigorously evaluated (Pearce and Turner, 2000). By fulfilling this role, geographical analysis can make an important contribution to the wider sustainability debate. When the issue at stake is as important as the long term viability of modern society, such knowledge gaps need to be speedily addressed. Only then will it be possible to determine with greater confidence whether the EU's current approach to environmental policy is capable of infusing the sustainability into development.

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Appendix 1 Contact letter for Preliminary Business Interviews

Date

Contact Name
Company Name
Address

Dear

Re: Producer Responsibility Obligations (Packaging Waste) Regulations 1997

I am a doctoral student with the Department of Geographical Sciences, University of Plymouth, currently conducting a research programme to examine the implementation of the EU Packaging Waste Directive and the UK Packaging Regulations.

The aim of the research is to explore the impact of the Directive and Regulations on businesses obligated by the UK legislation. Obviously, it is essential for me to obtain the opinions of as many affected industries as possible in order to gain a full understanding of these policies. It is my intention, therefore, to conduct a major survey of packaging producers in Britain and Germany. I am writing to you asking for assistance in providing preliminary information for this survey.

At present I am establishing an outline picture of how businesses in the UK perceive the Regulations. I would therefore be extremely grateful if you could spend a short time commenting on the questions raised below:

- Does your Company have specific plans for complying with the Packaging Regulations? If so, would it be possible to obtain a copy?
- What is the principal method employed by your Company to comply with the Regulations (compliance scheme or self-registration)?
- How are you managing the task of collecting packaging data in order to calculate your Company's recovery and recycling obligations?
- Can you please give brief comments on your Company perception of the Packaging Regulations in terms of i) the efficacy of the PRN system and the use of economic instruments and ii) the extent to which the EU Directive promotes packaging minimisation.

If you could please respond in writing to me at the above address, I would be extremely grateful. Please be assured that any responses you provide will be treated in accordance with the ethical research code of the University of Plymouth. This guarantees the anonymity, confidentiality and protection from harm of all research participants.

May I thank you for any help you can offer this research. I hope to hear from you soon.

Yours sincerely,

Ian Bailey
Department of Geographical Sciences

Appendix 2 Contact letter for Accredited Reprocessor Survey

Date

Contact Name
Company Name
Address

Dear

Re: Producer Responsibility Obligations (Packaging Waste) Regulations 1997 Research

I am currently conducting a research programme with the University of Plymouth, examining the implementation of the Packaging Waste Regulations. The research is a comparative study of the systems introduced within two European Union member states, the UK and Germany. It aims to evaluate their effectiveness in terms of (i) promoting sustainable waste management practices, and (ii) the impact of packaging regulation upon market systems and obligated producers.

As part of this research, I am contacting major companies in the packaging chain, including all compliance schemes and reprocessors. The purpose of this is to compile a profile how the reprocessing sector perceives the legislation. It would be of great assistance to the accuracy and objectivity of this study if you could supply the following information:

- Details of your current reprocessing centres and their capacity in terms of recycling and energy from waste (EfW), preferably also split by material type.
- Plans for new centres or expansions to existing recycling and EfW capacity.
- Details of partnerships with local authorities/industry to develop packaging collection schemes.
- The company's official position regarding the merits and problems of the PRN system.
- If you also have a publicly available operating prospectus, this would be extremely helpful.

I would like to stress that this research is bound by the University's code of ethical research. This ensures that all information remains strictly confidential, respondent interests are not compromised, and that participant anonymity outside the research is protected. I would further stress that the research is entirely academic in character and that the only objective is to produce an objective assessment of the recycling market for packaging waste.

I very much hope that you can assist this research and I look forward to hearing from you soon.

Yours sincerely,

Ian Bailey
Department of Geographical Sciences

Appendix 3a UK Survey Questionnaire

QUESTIONNAIRE CODE NUMBER

Please remember that all details will be treated as strictly confidential. For each question, tick the answer which best describes your organisation. Please read all options before selecting your answer. Please leave any questions you do not wish to answer blank.

SECTION A: IMPACT OF THE PACKAGING REGULATIONS UPON INDUSTRY

This section asks about the impact of the Packaging Regulations on your company. PLEASE TICK ONE BOX ONLY FOR EACH QUESTION UNLESS OTHERWISE SPECIFIED.

A1 Is your company aware of the Producer Responsibility Packaging Waste Regulations? IF NO, ANSWER SECTION E ONLY 1. Yes 2. No

A2 Does your company have legal responsibilities under the Packaging Regulations? IF NO, ANSWER SECTION E ONLY 1. Yes 2. No

A3 Which of the following methods has the company chosen in order to recovery and recycle its packaging?

1. Joined a compliance scheme 2. Organised packaging recovery independently

3. Other Please specify

A4 Does the company collect waste packaging at its own business sites? 1. Yes 2. No

A5 Does the company recover waste packaging from its customers? 1. Yes 2. No

A6 Approximately what percentage of the total packaging used by the company is collected directly (both at company sites and from customers)? %

A7 Which packaging materials does the company collect directly? PLEASE TICK THE BOX FOR EACH CATEGORY OF PACKAGING MATERIAL THE COMPANY COLLECTS

1. Paper/cardboard <input type="checkbox"/>	2. Glass <input type="checkbox"/>	3. Steel <input type="checkbox"/>
4. Aluminium <input type="checkbox"/>	5. Plastic <input type="checkbox"/>	6. Wood <input type="checkbox"/>

A8 How many of the company's employees are involved full time in managing its compliance with the Packaging Regulations?

A9 How many extra staff has the company employed in order to manage its compliance with the Regulations?

1. None 2. 1 or more If 1 or more, please specify

A10 Has the company employed any outside assistance to assist it achieve compliance with the Regulations, e.g. a consultancy? 1. Yes 2. No

SECTION B: PACKAGING POLICY

This section discusses the packaging policies adopted by your company. This section is split into three topics, Reduction, Reuse and Recycling. Please tick one box only for each question unless specified.

PACKAGING REDUCTION

B1 Has the company set targets for reducing the quantity of packaging it uses? **IF NO, GO TO QUESTION B5** 1. Yes 2. No

B2 What is the company's percentage reduction target between 1997 and 2001? %

B3 How is this reduction measured? **TICK EACH BOX WHICH DESCRIBES YOUR COMPANY'S STRATEGY FOR REDUCING PACKAGING**

- 1. A reduction in the **total** amount of packaging the company uses
- 2. A reduction in the weight of packaging for **each unit of product sold**

B4 Which packaging materials is the company reducing its usage of? **PLEASE TICK EACH BOX WHICH APPLIES**

- 1. Paper/cardboard
- 2. Glass
- 3. Steel
- 4. Aluminium
- 5. Plastic
- 6. Wood

REUSABLE PACKAGING

B5 Is the company using more reusable packaging than it did before the Packaging Regulations were introduced? 1. Yes 2. No

B6 Approximately what percentage (by weight) of packaging was reused in 1997?

B7 Approximately what percentage (by weight) will be reused by the year 2001?

B8 What types of reusable packaging materials does the company currently use? **PLEASE TICK EACH BOX WHICH APPLIES**

- 1. Paper/cardboard
- 2. Glass
- 3. Steel
- 4. Aluminium
- 5. Plastic
- 6. Wood

RECYCLING

B9 Is the company using more recycled packaging than it did before the Packaging Regulations were introduced? 1. Yes 2. No

B10 Approximately what percentage (by weight) of packaging used by the company was recycled in 1997?

B11 Approximately what percentage (by weight) will be recycled by the year 2001?

B12 What types of recycled packaging materials does your company currently use? **PLEASE TICK EACH BOX WHICH APPLIES**

- 1. Paper/cardboard
- 2. Glass
- 3. Steel
- 4. Aluminium
- 5. Plastic
- 6. Wood

SECTION C: FINANCIAL IMPACT OF THE PACKAGING REGULATIONS

C1 Which of the following categories of cost has the company incurred as a result of the Packaging Regulations? **PLEASE TICK EACH BOX WHICH APPLIES**

- a. Registration fee to a national enforcement agency
- b. Membership fee to a packaging compliance scheme or other similar organisation
- c. Unit or weight-based packaging recovery charges to a compliance scheme
- d. Deposits on non-returnable packaging produced by the company
- e. 'Green Dot' or similar charges
- f. Consultancy fee to packaging scheme
- g. Increased costs for waste collection
- h. Payment for packaging recovery certificates, e.g. PRNs
- i. Other, please specify

C2 What is the estimated cost to the company of complying with the Packaging Regulations for 1998?

C3 Please consider the following statements, and indicate on the scale given the company's opinion on each statement.

	+2 strongly agree	+1 agree	0 neither	-1 disagree	-2 strongly disagree
C3a The company will increase prices to customers to recover all the costs it incurs in complying with the Packaging Regulations	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3b The company will only recover part of these packaging costs from its customers	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3c The company will inform customers of the reason for these price increases	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3d The cost of the Regulations will encourage the company to change its policies on the use of packaging	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3e The company will only be able to comply with the minimum standards set by the Regulations	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3f The company aims to exceed the targets set by the Regulations	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3g The Regulations will encourage the company to reduce and reuse more of its packaging	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>

Section D: GOVERNMENT POLICY ON PACKAGING WASTE

For each statement, please tick the box for each statement which best describes your company's opinion on government policy on packaging. TICK ONE BOX ONLY FOR EACH QUESTION

		+2 strongly agree	+1 agree	0 neither	-1 disagree	-2 strongly disagree
D1	The recycling targets are too high for industry to achieve	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D2	The same recycling targets should apply in all EU states	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D3	The company has been unfairly discriminated against in the legislation	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D4	Voluntary targets from industry would have been a more effective way of promoting recycling than government legislation	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D5	The company could have achieved its recycling targets more easily if the government had not introduced charges for packaging waste	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D6	The government should specify that all packaging contains a set percentage of recycled material	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D7	Consumers should be taxed directly for packaging	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D8	More money should be spent on public education about recycling	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D9	The Packaging Regulations will achieve a cost-effective solution to Britain's packaging waste problems	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D10	The Regulations will produce worthwhile environmental benefits	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D11	In the company's opinion, packaging charges paid by the company are set on the basis of:					
a)	The cost of collecting and recycling packaging waste	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
b)	The full environmental impact of packaging production, use and disposal	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>

SECTION E: BUSINESS PROFILE

This section asks some details about your company. All details will be treated as strictly confidential. For each question, tick the answer which best describes your organisation.

E1 What is the annual turnover of your company? **TICK ONE BOX ONLY**

1. £0 - £4.99 million 2. £5 - £49 million 3. £50 - £99 million
4. £100 - £499 million 5. £500 - £999 million 6. £1,000 million +

E2 How many people does your company currently employ? **TICK ONE BOX ONLY**

1. 0-49 2. 50-99 3. 100-499
4. 500-999 5. 1,000-4,999 6. 5,000+

E3 What would you describe as the company's main area of activity (based on turnover)? **TICK ONE BOX ONLY**

1. Domestic trade 2. Import 3. Export
4. More than one category

E4 Which sector(s) of the Packaging Chain does your business primarily operate in? **Tick each box which applies**

1. Raw materials producer 2. Packaging manufacturer 3. Product manufacturer
4. Wholesaler 5. Retailer

E5 What weight (tonnes) of each of the following packaging materials did your company use in 1997? (full, not just obligation):

1. Paper/cardboard 2. Glass 3. Steel
4. Aluminium 5. Plastic 6. Wood

The paginating of this questionnaire has been changed to conform with thesis layout requirements. The content remains unaltered.

Appendix 3b German Survey Questionnaire

Fragebogen Nr.

Alle Antworten werden strikt vertraulich behandelt und nicht weitergegeben! Bitte kreuzen Sie für jede Frage die zutreffendste Antwort an. Bitte lesen Sie alle Antwortmöglichkeiten, bevor Sie Ihre Antwort markieren. Bitte überspringen Sie die Fragen, die Sie nicht beantworten wollen!

TEIL A: AUSWIRKUNGEN DER VERPACKUNGSVERORDNUNG AUF DIE UNTERNEHMEN

Dieser Abschnitt erhebt die Auswirkungen der Verpackungsverordnung auf Ihr Unternehmen. **BITTE KREUZEN SIE PRO FRAGE NUR EINE ANTWORT AN BZW. FÜLLEN SIE NUR EIN KÄSTCHEN AUS.**

A1 Kennt Ihr Unternehmen die Verpackungsverordnung? 1. Ja 2. Nein
FALLS NICHT WEITER MIT TEIL E!

A2 Hat Ihr Unternehmen rechtliche Verpflichtungen, die sich aus der Verpackungsverordnung ergeben? 1. Ja 2. Nein
FALLS NICHT WEITER MIT TEIL E!

A3 Wie sammelt und recycelt Ihr Unternehmen den anfallenden Verpackungsabfall??

1. Teilnahme an einem bestehenden Entsorgungskonzept (z.B. DSD)

2. Eigenes Entsorgungskonzept

3. Anderes Bitte spezifizieren

A4 Sammelt Ihr Unternehmen Verpackungsabfälle an den eigenen Standorten? 1. Ja 2. Nein

A5 Sammelt Ihr Unternehmen den Verpackungsabfall Ihrer Kunden? 1. Ja 2. Nein

A6 Welcher Anteil des gesamten Verpackungsmaterials wird durch Ihr Unternehmen wieder eingesammelt (an den Standorten und von Kunden)? %

A7 Welche Verpackungsmaterialien sammelt Ihr Unternehmen? **BITTE KREUZEN SIE NUR DIE MATERIALIEN AN, DIE IHR UNTERNEHMEN SELBST SAMMELT**

1. Papier/Pappe 2. Glas 3. Eisen/Stahl/Blech

4. Aluminium 5. Plastik 6. Holz

A8 Wieviele Ihrer Mitarbeiter sind unmittelbar mit der Regelung der Anforderungen der Verpackungsverordnung befaßt?

A9 Wieviele Arbeitskräfte hat Ihr Unternehmen zusätzlich eingestellt, um den Anforderungen der Verpackungsverordnung zu genügen?

1. Keine 2. 1 oder mehr Falls 1 oder mehr, wieviele?

A10 Hat Ihr Unternehmen Unterstützung oder Beratung von anderen Unternehmen angenommen (z.B. Unternehmensberatungen), um den Anforderungen der Verordnung zu genügen? 1. Ja 2. Nein

TEIL B: UMSETZUNG DER VERPACKUNGSVERORDNUNG

Dieser Abschnitt beschäftigt sich mit der Umsetzung der Verpackungsverordnung in Ihrem Unternehmen und ist in die drei Teile Abfallverringerung, -wiederverwertung und -recycling aufgeteilt. Bitte kreuzen Sie nur eine Antwort pro Frage an.

VERRINGERUNG DER VERPACKUNGSMENGE

B1 Hat sich Ihr Unternehmen konkrete Ziele zur Verringerung des Verpackungsmaterials gesetzt? **FALLS NICHT WEITE MIT FRAGE B5** 1. Ja 2. Nein

B2 Wieviel Prozent des Verpackungsmaterials soll zwischen 1997 und 2001 eingespart werden?

B3 Wie soll diese Einsparung erreicht werden? **BITTE KREUZEN SIE DIE ANTWORT AN, DIE DIE STRATEGIE IHRES UNTERNEHMENS BESCHREIBT**

1. Eine Verminderung des Gesamtaufkommens an Verpackungsmaterial
2. Eine Verminderung des Verpackungsgewichts des verkauften Produkts

B4 Welche Verpackungsmaterialien versucht Ihr Unternehmen im Verbrauch zu reduzieren? **BITTE KREUZEN SIE DIE ZUTREFFENDEN ANTWORTEN AN**

1. Papier/Pappe 2. Glas 3. Eisen/Stahl/Blech
4. Aluminium 5. Plastik 6. Holz

WIEDERVERWERTBARE VERPACKUNGEN

B5 Verwendet Ihr Unternehmen heute mehr wiederverwertbare Verpackungen als vor Einführung der Verpackungsverordnung? 1. Ja 2. Nein

B6 Welcher Anteil des Verpackungsmaterials (Gewicht) wurde 1997 wiederverwendet?

B7 Welcher Anteil des Verpackungsmaterials (Gewicht) soll im Jahre 2001 wiederverwendet werden?

B8 Welche Arten von wiederverwertbaren Verpackungen nutzt Ihr Unternehmen schon heute? **BITTE KREUZEN SIE DIE ZUTREFFENDEN ANTWORTEN AN**

1. Papier/Pappe 2. Glas 3. Eisen/Stahl/Blech
4. Aluminium 5. Plastik 6. Holz

RECYCLING

B9 Nutzt Ihr Unternehmen heute mehr recyceltes Verpackungsmaterial als vor Einführung der Verpackungsverordnung? 1. Ja 2. Nein

B10 Welcher Anteil des Verpackungsmaterials (Gewicht) wurde 1997 recycelt

B11 Welcher Anteil des Verpackungsmaterials (Gewicht) wird im Jahre 2001 recycelt werden?

B12 Welche Arten von recycelten Verpackungsmaterialien nutzt Ihr Unternehmen schon heute? **BITTE KREUZEN SIE DIE ZUTREFFENDEN ANTWORTEN AN**

- | | | | | | |
|-----------------|--------------------------|------------|--------------------------|----------|--------------------------|
| 1. Papier/Pappe | <input type="checkbox"/> | 2. Glas | <input type="checkbox"/> | 3. Stahl | <input type="checkbox"/> |
| 4. Aluminium | <input type="checkbox"/> | 5. Plastik | <input type="checkbox"/> | 6. Holz | <input type="checkbox"/> |

Teil C: FINANZIELLE AUSWIRKUNGEN DER VERPACKUNGSVERORDNUNG

C1 Welche der nachfolgend aufgelisteten Kosten entstanden Ihrem Unternehmen im Zusammenhang mit der Verpackungsverordnung? **BITTE KREUZEN SIE DIE ZUTREFFENDEN ANTWORTEN AN:**

- | | |
|---|--------------------------|
| a. Registrierungsgebühren an eine nationale Umsetzungsbehörde | <input type="checkbox"/> |
| b. Mitgliedsgebühren in einem Entsorgungskonzept | <input type="checkbox"/> |
| c. Stück- oder gewichtsmäßige Gebühren zur Abfallsammlung | <input type="checkbox"/> |
| d. Anzahlungen für Verpackungsmaterial, das nicht zurückgegeben werden kann | <input type="checkbox"/> |
| e. "Grüner Punkt" o.ä. | <input type="checkbox"/> |
| f. Beratungskosten in Zusammenhang mit Verpackungen | <input type="checkbox"/> |
| g. Gestiegene Kosten der Abfallsammlung | <input type="checkbox"/> |
| h. Zahlungen für Wiederverwertungs- oder Recyclingszertifikate | <input type="checkbox"/> |
| i. andere, bitte spezifizieren | <input type="text"/> |

C2 Wie hoch waren 1998 die geschätzten Kosten der Verordnung in Ihrem Unternehmen?

C3 Bitte lesen Sie die folgenden Aussagen und geben Sie die Haltung Ihres Unternehmens zu den Aussagen an

	+2 stimme voll zu	+1 stimme zu	0 weder- noch	-1 stimme nicht zu	-2 stimme absolut nicht zu
C3a Das Unternehmen wird die Verbraucherpreise erhöhen, um die zusätzlichen Kosten der Verpackungsverordnung aufzufangen	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3b Das Unternehmen wird nur einen Teil dieser Kosten von seinen Kunden wiedererhalten	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3c Wir werden unsere Kunden die Gründe der Preiserhöhung informieren	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>

C3d	Die zusätzlichen Kosten der Verpackungsverordnung wird das Unternehmen dazu ermuntern, seine Verpackungspolitik zu überdenken	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3e	Das Unternehmen wird nur die Minimalanforderungen der Verpackungsverordnung erfüllen können	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3f	Das Unternehmen hat sich das Ziel gesetzt, die Ziele der Verordnung voll zu erreichen	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
C3g	Die Verordnung wird die Firma dazu bewegen weniger Verpackungsmaterial zu nutzen und mehr davon wiederzuwerten, als es sonst getan hätte	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>

Teil D: VERPACKUNGSPOLITIK

Bitte lesen Sie die Aussagen und geben Sie die Meinung Ihres Unternehmens zur Verpackungspolitik an. BITTE KREUZEN SIE NUR EIN KÄSTCHEN PRO FRAGE AN.

		+2 stimme voll zu	+1 stimme zu	0 weder- noch	-1 stimme nicht zu	-2 stimme absolut nicht zu
D1	Die Recyclingziele sind für Unternehmen zu hoch angesetzt worden	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D2	In allen EU-Staaten sollten dieselben Recyclingziele gelten	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D3	Das Unternehmen wurde durch die Gesetzgebung unfair benachteiligt	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D4	Freiwillige Zielsetzungen durch die Unternehmen wären effizienter gewesen als gesetzgeberische Maßnahmen	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D5	Das Unternehmen hätte seine Recyclingziele rascher erreicht, wenn die Regierung nicht Gebühren auf Verpackungsabfälle erhoben hätte	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D6	Die Regierung sollte festlegen, daß Verpackungsmaterialien einen bestimmten Anteil an Recyclingmaterial enthalten sollten	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D7	Endverbraucher sollten direkt für Verpackungen steuerlich belastet werden	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D8	Für öffentliche Informationskampagnen zum Thema Recycling sollte mehr Geld ausgegeben werden	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D9	Die Verpackungsverordnung wird eine kostenwirksame Lösung der Verpackungsabfallprobleme mit sich bringen	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>
D10	Die Verpackungsverordnung wird meßbare Verbesserungen im Umweltbereich nach sich ziehen	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>	<input type="checkbox"/>

D11 Nach Ansicht des Unternehmens basieren die Verpackungsgebühren, die das Unternehmen entrichtet, auf:

- a) Den tatsächlichen Kosten des Sammelns und Recyclings der Verpackungsabfälle
- b) Den Kosten der Produktion, Nutzung und Entsorgung von Verpackungsmaterialien einschließlich der dabei entstehenden Umweltbelastungen.

TEIL E: PROFIL DES UNTERNEHMENS

Dieser Abschnitt erhebt einige Details Ihres Unternehmens. Alle Einzelheiten werden strikt vertraulich behandelt und nicht weitergegeben oder veröffentlicht. Bitte kreuzen Sie die auf Ihr Unternehmen zutreffendste Antwort an.

E1 Wie hoch ist der Jahresumsatz Ihres Unternehmens? **BITTE NUR EINE ANTWORT**

1. DM 0-14,99 Mio 2. DM 15-139 Mio
3. DM 140-274 Mio 4. DM 275 Mio-1,379 Mrd
5. DM 1,380 Mrd-2,749 Mrd 6. DM über 2,750 Mrd

E2 Wieviele Mitarbeiter beschäftigt Ihr Unternehmen gegenwärtig? **BITTE NUR EINE ANTWORT**

1. 0-49 2. 50-99 3. 100-499
4. 500-999 5. 1.000-4.999 6. über 5.000

E3 Wo würden Sie das Hauptgeschäftsfeld (basierend auf Umsatzzahlen) Ihres Unternehmens sehen?

1. Binnenhandel 2. Import 3. Export
4. > 1

E4 In welchen Teilen der Verpackungskette ist Ihr Unternehmen vorwiegend involviert? **MEHRERE ANTWORTEN MÖGLICH**

1. Produktion von Rohmaterialien 2. Herstellung von Verpackungen
3. Herstellung von Produkten 4. Großhandel
5. Einzelhandel

E5 Wieviel Tonnen der nachfolgend aufgelisteten Verpackungsmaterialien nutzte Ihr Unternehmen 1997?

1. Papier/Pappe 2. Glas 3. Eisen/Stahl/Blech
4. Aluminium 5. Plastik 6. Holz

The paginating of this questionnaire has been changed to conform with thesis layout requirements. The content remains unaltered.

Appendix 4 Producer Survey Pilot Feedback Form

Producer Responsibility Obligations (Packaging Waste) Regulations 1997 Feedback Form

1. **Length of Questionnaire** - do you feel that business managers would take the time to complete and return the questionnaire? Is it too long or too short?

2. **Question Content** - Are the questions and instructions clear and easy to understand? Were any points not clear? Do you find the questionnaire layout format helpful?

3. **Factual accuracy** - Do you feel all the questions are relevant in your experience of the UK Packaging Regulations? Are there any important features of the legislation which are not explored within the questionnaire?

4. **Sensitive Information** - Is there any information requested in the questionnaire that you would not feel happy about providing? Do you feel the assurance of confidentiality is adequate?

Appendix 5a Producers Survey UK Contact Letter

The Managing Director
Company Name
Address

Date

Dear Sir or Madam

Re: Producer Responsibility Packaging Waste Regulations

I am a doctoral student with the University of Plymouth and am currently conducting a research programme examining the implementation of the European Union Packaging Waste Directive. Results of the study will be published in the UK by the research's co-sponsor, the Institute of Wastes Management.

The aims of the research are, first, to explore the impact of the Directive on national packaging legislation, and second, to evaluate the methods used to implement the Directive. As you are aware, environmental legislation is having an increasing effect on industry and it is therefore essential that new measures are able to achieve their objectives without undermining industrial competitiveness.

Obviously, if the study is to evaluate environmental policies meaningfully, the views of industry should form a core part of its results. I am therefore conducting a survey of 1,800 companies in Britain and Germany to assess the impact of packaging regulations within these states. If sufficient companies respond to the survey, the research will be able to provide an authoritative picture of the Packaging Directive's impact on industry and could help guide the development of future environmental policies.

I am therefore writing to ask if you could spend a short time responding to the enclosed questionnaire and returning it in the Freepost envelope provided. The questionnaire should take about 30 minutes to complete and will provide invaluable information for this study. May I also assure you that all information will be treated as strictly confidential and that the identity and details of participating companies will not be disclosed or identifiable at any point in the research results.

I would like to thank you for your assistance in this work, and look forward to hearing from you soon.

Yours faithfully,

Ian Bailey
Department of Geographical Sciences
Enc.: Packaging Waste questionnaire, SAE

Appendix 5a Producers Survey German Contact Letter

Der Generaldirektor
Company Name
Address

Date

Europäische Direktive über Verpackung und Verpackungsabfall

Sehr geehrte Damen und Herren,

als Doktorand am Geographischen Institut der University of Plymouth, England führe ich eine Untersuchung zu den Aufwirkungen der Europäischen Direktive über Verpackung und Verpackungsabfall in den Mitgliedsstaaten der EU durch. In Zusammenhang mit diesem Forschungsvorhaben, das übrigens durch das britische Institute of Waste Management mitfinanziert wird, möchte ich Sie um Ihre Mithilfe bitten.

Die Zielsetzungen des Forschungsvorhabens umfassen zum einen die Analyse der Auswirkungen der Direktive auf die Verpackungs- und Verpackungsabfallgesetzgebungen in den Mitgliedsstaaten und zum anderen eine Bewertung des Erfolgs von marktorientierten Implementierungsmethoden, dargestellt am Beispiel der Direktive. Wie Sie sicher wissen, beeinflußt die Umweltgesetzgebung in zunehmenden Maße die Tätigkeit von Unternehmen aller Sparten. Es ist daher wichtig, daß neu eingeführte Maßnahmen nicht nur ihre umweltpolitischen Ziele erreichen, sondern auch die Konkurrenzfähigkeit der Unternehmen nicht negativ beeinflussen.

Wenn diese Untersuchung tatsächlich die Umsetzung der Direktive bewerten soll, ist es unerlässlich, daß die Meinungsbildung innerhalb der betroffenen Unternehmen das Kernstück der Untersuchung bildet. Aus diesem Grunde führe ich in England und Deutschland eine Umfrage zur Auswirkung von Verpackungs- und Abfallentsorgungsrichtlinien durch. Am Ende der Studie sollte dann ein aussagekräftiges Bild der Auswirkungen solcher Verordnungen auf die Unternehmenstätigkeit in den genannten Ländern stehen; anhand der Ergebnisse einer solchen Studie könnten dann zukünftige umweltpolitische Maßnahmen bereits vor ihrem Inkrafttreten auf wirtschaftspolitische Vertretbarkeit geprüft werden.

Ich möchte Sie daher bitten, den beiliegenden Fragebogen auszufüllen und mit Hilfe des beiliegenden adressierten, frankierten Umschlags an mich zurückzusenden. Das Ausfüllen des Fragebogens selbst dauert etwa 30 min, liefert aber für das Projekt unerlässliche Informationen. Ich kann Ihnen an dieser Stelle versichern, daß alle Informationen selbstverständlich vertraulich behandelt werden und daß eine Identifizierung der Unternehmen anhand der Fragebögen unmöglich sein wird.

Ich möchte Ihnen schon jetzt für Ihre Mithilfe danken und hoffe, bald von Ihnen zu hören.

Mit freundlichen Grüßen

Ian Bailey
Department of Geographical Sciences
Anlage: Fragebogen, frankierter Rückumschlag

Appendix 6a Producers Survey UK Reminder Letter

Date

The Managing Director
Company Name
Address

Dear Sir or Madam

RE: PRODUCER RESPONSIBILITY (PACKAGING WASTE) REGULATIONS SURVEY

You may recall that I wrote to you in early January, requesting your company's participation in a survey to assess the impact of the Packaging Regulations on your organisation's policies towards packaging waste management.

The study, which is being conducted as part of my doctoral thesis, aims to compare the effects of the different forms of packaging legislation introduced within the Member states of the European Union. It then seeks to assess their ability to produce worthwhile environmental benefits in a manner which does not undermine industrial competitiveness.

According to my records, I have not yet received a reply from your company. I understand that my request may have come at a busy time and that workload pressures are great at all times of the year. However, if the study is to reflect the impact of the Regulations on industry accurately, it is important that the views of as many affected companies as possible are taken into account.

Since I last wrote, the European Commission is now itself seeking studies on the effectiveness of packaging regulations in the Member States. It is hoped that work from this survey may be submitted as part of this study and therefore that the impact of packaging legislation on industry can be reported directly to the Commission to help inform future decisions on waste management issues.

I am therefore writing to ask again if you could spend a short time responding to the enclosed questionnaire and returning it in the Freepost envelope provided. The questions should take about 30 minutes to complete. May I also again reassure you that all information provided will be treated, in accordance with the University's code of ethical research, as strictly confidential and that the identity and details of participating companies will not be disclosed or identifiable at any point in the research results.

I very much hope you are able to assist this study, and I look forward to hearing from you soon.

Yours faithfully

Ian Bailey
Department of Geographical Sciences
Enc: Packaging Waste questionnaire, Freepost envelope

Appendix 6b Producers Survey German Reminder Letter

Date

Der Generaldirektor
Company Name
Address

Betrifft: Europäische Direktive über Verpackung und Verpackungsabfall

Sehr geehrte Damen und Herren,

Im Rahmen meiner Doktorarbeit am Geographischen Institut der University of Plymouth, GB, führe ich eine Untersuchung zur Europäischen Direktive über Verpackung und Verpackungsabfall in den Mitgliedsstaaten der EU durch. Die Zielsetzungen des Forschungsvorhabens umfassen (i) die Analyse der Auswirkungen der Direktive auf die Verpackungs- und Verpackungsabfallgesetzgebungen in den Mitgliedsstaaten und (ii) eine Bewertung des Erfolgs von marktorientierten Methoden der Implementierung, dargestellt am Beispiel der Direktive. Da die Umweltgesetzgebung in zunehmendem Maße die Tätigkeit von Unternehmen beeinflusst. Ist es wichtig, daß neu eingeführte Maßnahmen nicht nur ihre umweltpolitischen Ziele erreichen, sondern auch die Konkurrenzfähigkeit der Unternehmen nicht negativ beeinflussen.

Sicher erinnern Sie sich daran, daß ich Ihrem Unternehmen einen Fragenbogen zu dem oben genannten Vorhaben zugeschickt habe. Meinen Unterlagen entnehme ich, daß noch ich keinen Rückbrief aus Ihrem Betrieb erhalten habe. Es ist mir verständlich, daß Sie meine Anfrage eventuell zu einer für Ihr Unternehmen ungünstigen Zeit erreichte und einige der gestellten Fragen einen relativ sensiblen Geschäftsbereich betreffen. Allerdings ist es für genaue Aussagen über Auswirkungen der Verpackungsverordnung unerlässlich so viele Eindrücke Deutsche Unternehmen wie möglich repräsentieren zu können.

Ich bitte Sie daher nochmals zu erwägen, ob Sie eine kurze Weile zur Beantwortung des beigefügten Fragebogens aufbringen können. Ich habe den Fragebogen verkürzt damit er schneller auszufüllen ist.

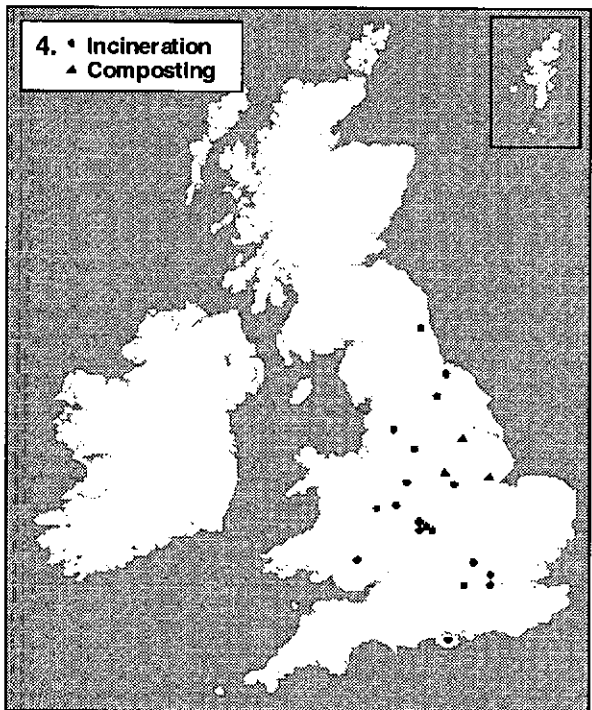
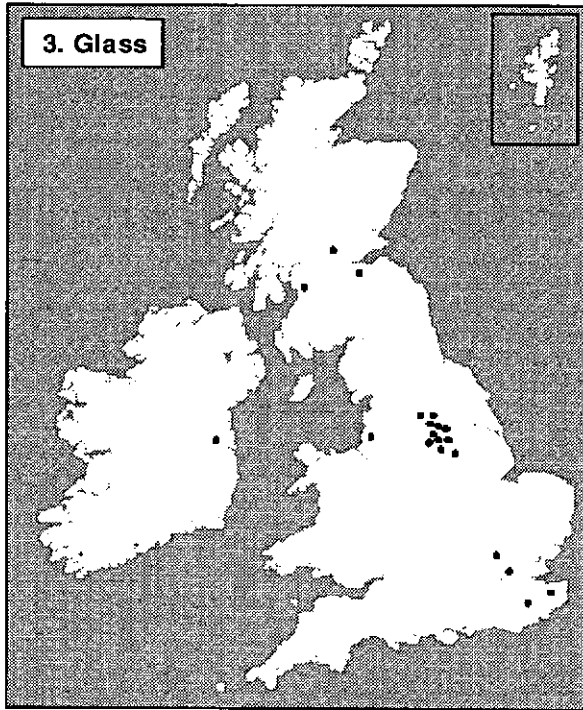
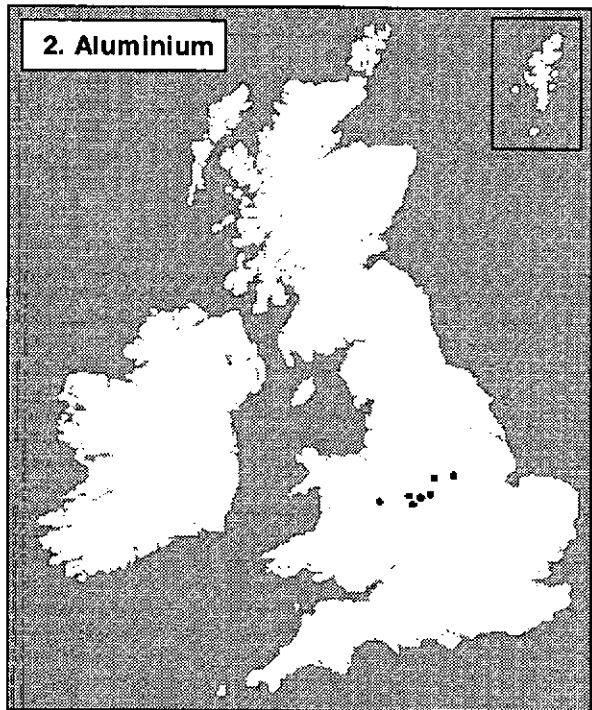
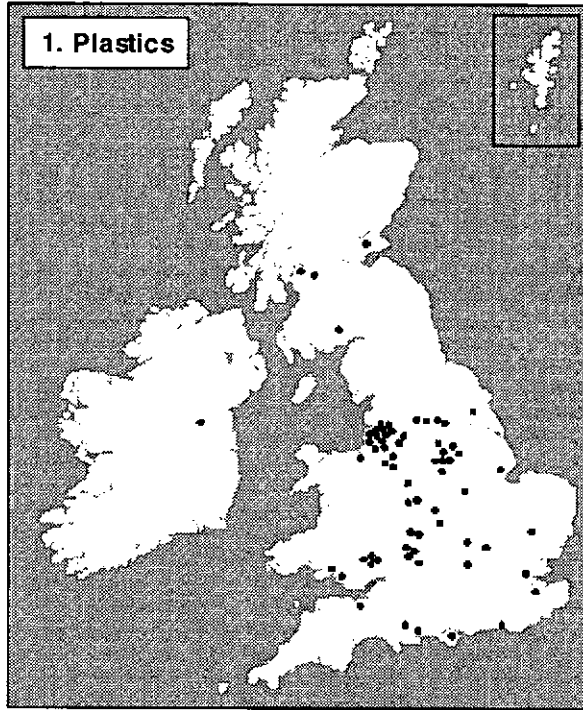
Ich kann Ihnen an dieser Stelle nochmals versichern, daß alle Informationen selbstverständlich vertraulich behandelt werden und daß eine Identifizierung der Unternehmen anhand der Fragebögen unmöglich sein wird.

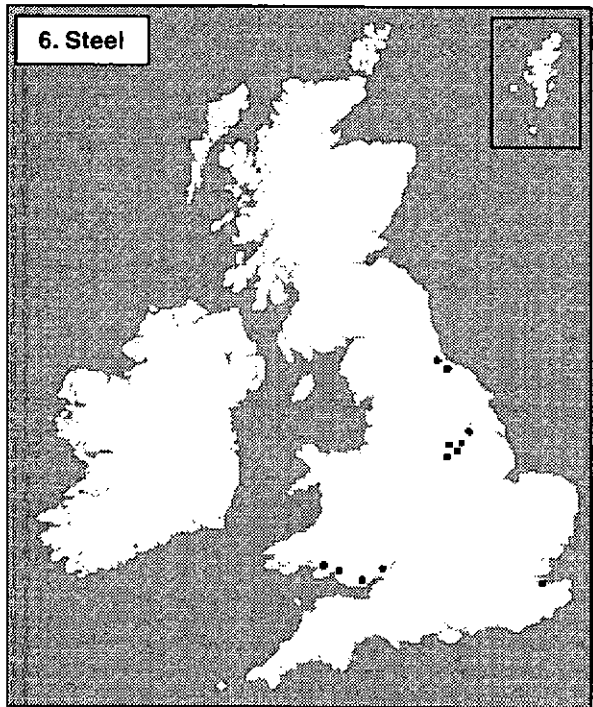
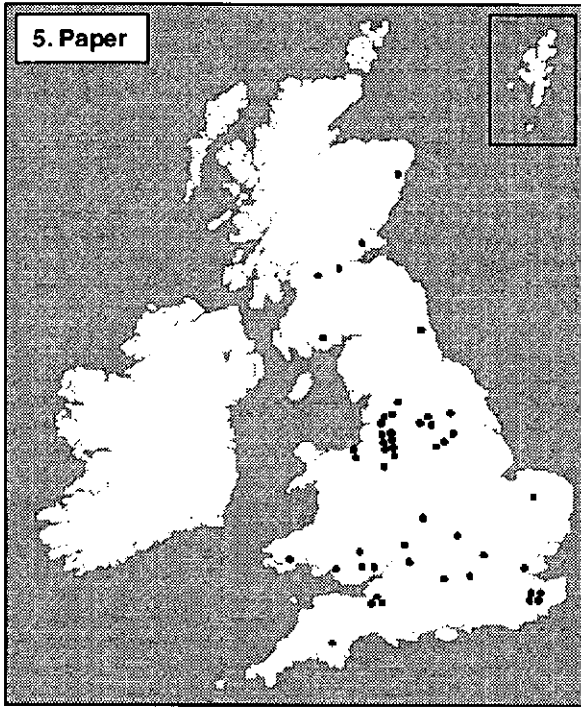
Ich möchte Ihnen schon jetzt für Ihre Mithilfe danken und hoffe, bald von Ihnen zu hören.

Mit freundlichen Grüßen

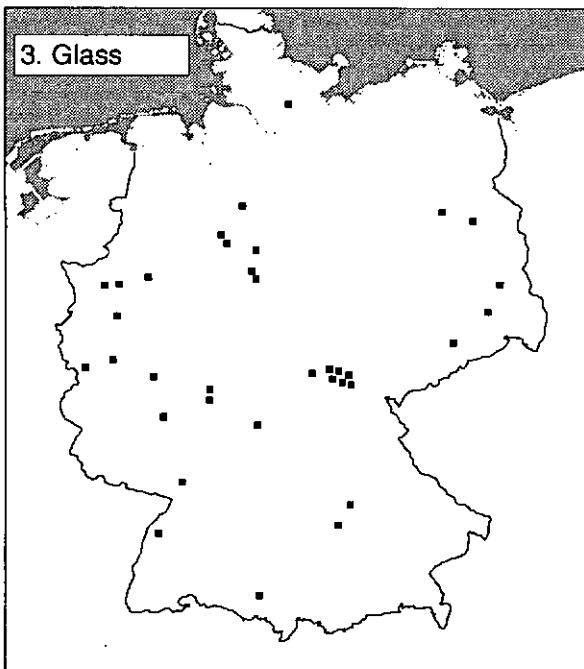
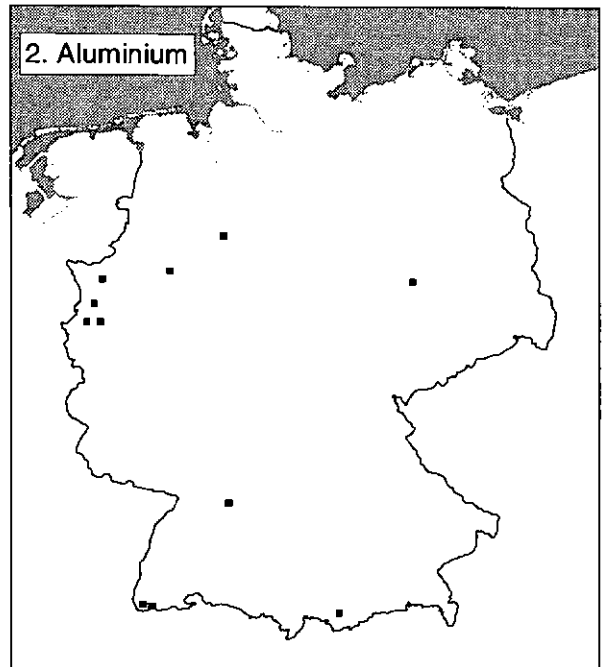
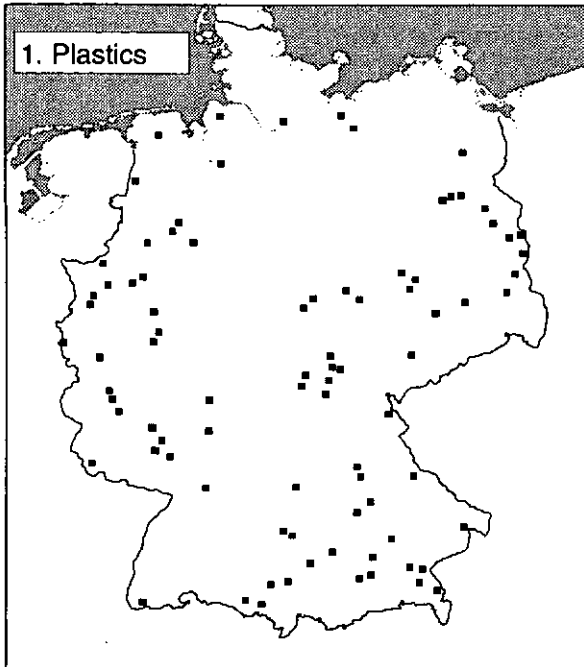
Ian Bailey
Department of Geographical Sciences
Anlage: Fragebogen, frankierter Rückumschlag

Appendix 7a Distribution of Reprocessing Facilities, Britain





Appendix 7b Distribution of Reprocessing Facilities, Germany



4. No data for Incineration and Composting

