Sustainability Analysis of a Transport System: The UK Car Fleet, 1995-2005

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To my parents,

to whom I owe more than words can express.

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Abstract

A sustainability analysis was performed on the UK car fleet in 1995 and 2005, which showed that the sustainability of the car fleet improved by 13% over the time period. The Process Analysis Method was applied to generate a reliable, consistent, and sufficient indicator set describing the sustainability impact of a transport system. As well as identifying conventional impacts, this analysis identified that the quality of the mobility produced has an important impact on sustainability. Quality affects how well mobility meets human needs, but also influences *user choices,* which drive system change. It is suggested that the indicator set that was produced should form the basis for future sustainability assessments. A study was carried out on the UK car fleet in 1995 and 2005 to test the indicator set and to investigate changes in environmental, human/social and economic sustainability over the period. Distance travelled increased, driven by a 28% increase in fleet size, which counteracted increases in fuel efficiency. Weight also increased, driven by increasing vehicle size, and additional safety and comfort-related features, increasing embodied energy and emissions. However, direct emissions of non-CO_2 pollutants reduced due to increasingly stringent regulation, as did waste to landfill. CO_2 emissions increased by 11% overall. The reduction in emissions had a positive impact on health, with emission related mortality reducing by 27%. Casualties from collisions involving cars reduced by 13%. Road accidents were found to represent an impact on mortality three times greater than emissions. The indicators of service quality showed a drop in average speed and reliability (from both the network and vehicle perspectives) but an increase in comfort and vehicle safety. The increase in fleet size and new registrations drove growth in the retail and maintenance sector, which saw growth of 25% in gross value added. The opposite happened to the manufacturing sector, which shrank by 26%, due to stagnant domestic production and an increasingly competitive market affecting profit margins. An indicator aggregation was performed to summarise changes in the sustainability of car-based mobility. The indicators were normalised to a percentage change, and weighted according to their monetary value.

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Chapter 1

Introduction and Literature Review

Increasing awareness of the problems caused by the dominant economic model of development has led to interest in adopting a more sustainable way of life at all levels of society. This leads to the need for an independent quantitative means of assessing what is sustainable and progress towards sustainability. The most often quoted figure for sustainability tends to be emissions levels, especially greenhouse gases. However, this emphasis on emissions fails to fully address the impact of a system on sustainable development, which is not limited to CO_2. A multi-component complex system cannot be described with a single number or reference point, but by a carefully chosen set of indicators and metrics. A variety of indicator frameworks are in use, such as those generated by the UK Government's Department for Food and Rural Affairs (DEFRA, 2008), or the reporting framework suggested by the Global Reporting Initiative (GRI, 2006). Reliance on too few indicators can lead to perverse incentives for action and undesirable outcomes. Thus, careful testing of the method used to select the indicator set is essential to ensure that it truly reflects the requirements for sustainability.

Road transport has seen significant growth in recent years, and increased awareness of transport issues such as emissions and congestion has raised concerns over its sustainability. It thus provides an interesting and useful case study with which to test new methods of performing sustainability assessment. A holistic approach will offer a set of indicators and metrics capable of fully describing its impact on sustainability, and assess system evolution. This is an important question in terms of evaluating the effectiveness of levers to improve the sustainability of the system – such as regulations or government policies – and in identifying what factors are having the greatest influence.

1.1. Sustainability Definitions

Defining sustainability is the first step towards being able to assess or measure it. The context in which one is trying to judge sustainability strongly influences the choice of indicators, or more generally the means of assessment. Furthermore, many definitions are vague and open to interpretation. They are especially unclear on how to implement sustainability and embody the concept into a framework that allows systems to be critically assessed and clear targets for their evolution to be defined.

The most quoted definition of sustainability (or more specifically sustainable development) is that of Brundtland (Brundtland, 1987) which states that:

> *"Sustainable development is development that meets the needs of the current generations without compromising the ability of future generations to meet their own needs."*

A key factor of this definition is that it is broad enough to encompass the three main aspects of sustainability, namely social, economic and environmental sustainability. It relays the underlying concept of sustainability, the idea of intergenerational equity – ensuring the planet remains habitable for future generations. However, in practical terms it remains open to interpretation. The most obvious point of contention is how to predict the needs of future generations: advances in technology, foreseen or unforeseen natural events, changes in society, could change these 'needs'. This potentially leads to conflict. A typical example is whether it is acceptable to use agricultural land for biofuels (preserving fossil fuels for future generations) when that land could be used for growing food for people starving in developing countries.

The Brundtland definition, and the surrounding report, was the result of the first large international effort to examine the global environmental issues in conjunction with issues related to human development. It had a strong influence on the 1992 United Nations Conference on Environment and Development (UNCED) held in Rio de Janeiro, Brazil which produced Agenda 21, an action plan for implementing sustainability on a local, national and global scale, and has continued to strongly influence the sustainability debate. Although the Brundtland definition has become the most established, there are other interpretations or approaches to sustainability. A few have been selected here for their different insights into the issue. A number of definitions of sustainability accommodate the possibility that a sustainable system can evolve, such as that of Clayton and Radcliffe (1997) who consider a “systems model” of sustainability, whereby the planetary system exists at a point in ‘phase space’ corresponding to its current state. As various elements interact (e.g. humans and their environment) the planetary system changes state, and hence moves to a different point in phase space. Sustainability, from the human perspective, is defined as staying within the region of phase space in which humanity can survive. The more humanity indulges in unsustainable practices, such as consuming a critical resource, the closer the planetary system gets to the edge of this region. As this occurs, life becomes harder for humanity. Moreover, the faster the system moves through phase space, the more dangerous the change becomes, as it allows less time for the inhabitants of the system to adapt to their new environment. This approach accepts change within certain boundaries – there is no fixed state that needs to be maintained. In this it echoes Brundtland’s emphasis on development, but does not emphasise positive change or equity in the same way.

The idea of a sustainable system being in constant evolution can in fact be considered a criterion for sustainability, as put forward by Dovers and Handmere (1992), who define sustainability as:

> *"[...]the adaptability of human, natural or mixed systems to withstand or adapt to endogenous or exogenous change indefinitely. Sustainable development is therefore a pathway of deliberate change and improvement which maintains or enhances this attribute of the system, while answering the needs of the present population."*

Rather than focus on resource use to ensure that future generations are able to meet their needs, this approach states that a system must be adaptable to change. This ensures that as problems arise, or resources become scarce, the system is able to adapt and change in such a fashion as to survive. Change becomes a feature of sustainability.

Some researchers have abandoned the idea of elaborating a comprehensive and universally applicable definition for what constitutes sustainability, focusing instead on developing working definitions through stakeholder involvement. Bell and Morse (1999) assess sustainability on a case by case basis by means of a collaborative, 'soft systems' method featuring input from all the different stakeholders or stakeholder groups, to ensure that the project shares a common vision of sustainability. This method is cyclical rather than linear – the requirements for sustainability are constantly re-evaluated in a 'continuous learning cycle'. The reasoning behind this approach is that definitions are arbitrary, and that a collaborative method is more likely to yield a useful result. However it can be argued that the selection of stakeholders is just as arbitrary in terms of influencing the result. Not all stakeholders, such as future generations, can be included.

A number of definitions choose to focus on the consideration of resources and resource flow through the ecosystem. For instance, Rennings and Wiggering (1997) state three rules for the sustainable use of natural resources:

- Harvest rates of renewable resources should not exceed regeneration rates

- Waste emissions should not exceed the relevant assimilative capacities of ecosystems
- Nonrenewable resources should be exploited in a quasi-sustainable manner by limiting their rate of depletion to the rate of creating renewable substitutes

Whilst this is not a complete definition of sustainability, in that it only focuses on resource use, it has some interesting features, such as the approach towards non-renewable resources. A similar definition is suggested by Holmberg *et. al* (1996) who state that a system is sustainable if and only if:

- Substances extracted from the lithosphere do not systematically accumulate in the ecosphere.
- Society-produced substances do not systematically accumulate in the ecosphere.
- The physical conditions for production and diversity within the ecosphere do not systematically deteriorate.
- The use of resources is efficient and just with respect to meeting human needs.

Daly (1991) provides four operational principles of sustainability which are very close to those stated in the definitions above, but with a slightly different emphasis: he states that "*the main principle is to limit the human scale throughput to a level which, if not optimal, is at least within the carrying capacity and therefore sustainable*" and that "*technological progress should be efficiency-increasing rather than throughput-increasing*". This brings up the crucial issue of scale (and by extension, growth) and sustainability – the larger the system becomes, the more it stretches the carrying capacity of its environment.

The resource-centric definitions all emphasise minimising the impact of humans on the environment as the way towards sustainability, without really considering social aspects.

The Natural Step, a not-for-profit organisation concerned with research and education on sustainable development, attempt to remedy this by including in their four 'principles of sustainability' (which largely resemble those of Holmberg *et al.*) a clause that states "*in that society, people are not subject to conditions that systemically undermine their capacity to meet their needs*". Gudmundsson and Hojer (1996) expand on this with:

> *"[..] a conception of sustainable development assessment where sustainability and development represent different dimensions. Sustainability refers to criteria for long-term stability of the social system, relevant to future generations, while development is the perceptible improvement of the quality of human life"*

The development aspect also includes equity in terms of distribution of quality of life. This conception of sustainable development seems particularly well suited to the assessment of human systems, which are all fundamentally created to improve quality of life.

Industrial Ecology could also be seen as a means of defining and implementing sustainability, again with a focus on resource flows. The key concept behind it is to use non-human, natural ecosystems as models for industrial activity (Ayres 2002). Graedel and Allenby (1994) proposed a framework for classifying ecosystems. They define an ideal system (known as a type 3 system – type 1 and 2 systems being less sustainable systems) that consists of a closed materials loop driven by an input of energy from a sustainable source (usually the sun). The crucial aspect of this ideal system is that it is closed – there are no emissions or material inputs. All waste materials are converted into useful primary materials through the input of energy. This means that there is no impact on the environment.

The interesting element of IE is that it provides a strong theoretical basis for practical assessment and implementation within 'real' industrial systems. Ayres (2002) discusses 'industrial ecosystems'; Jackson and Geyer (2004) discuss the use of supply loops; here, wastes are used as primary resources to secondary processes, and end-of-life products are recycled or reused, so that there is no net waste. Jackson and Geyer note however that if the primary products are not designed for recycling or reuse from the outset the process becomes far less efficient. This is applicable to all areas of industrial ecology – if no thought is given to what uses can be made of the wastes whilst designing the primary process opportunities might be missed to design a slightly different primary process with more useful wastes.

Based as it is on an analogy to natural ecosystems, Industrial Ecology has not historically explicitly addressed the economic and social aspects of sustainability (Ehrenfeld, 2007). Some social benefit is implicit in the idea that resources are consumed to produce useful products, and in the reduction of negative environmental impacts, but this is not the same as making development a stated goal. The focus is on maximum efficiency, and reduction of all wastes and consumption. No consideration is given to the quality of a system's contribution to society. Using IE as a definition of sustainability may be pushing the concept beyond its capabilities – IE was designed as a means of making industrial processes cleaner and less resource intensive by modelling them on natural systems, and as such is perhaps best seen as a practical response to the issues raised by sustainable development.

Sustainability does not imply a fixed state; the earth is constantly changing and evolving. Consequently, goals and targets must reflect this by being flexible and adapting to changing circumstances. Bell and Morse's idea of continuously updating the indicators to reflect lessons learned and changes in the environment fits with this view of things. However, Bell and Morse believe that sustainability can only be defined by stakeholder consensus, which raises some issues, such as how to guarantee that all stakeholders are included (especially future generations), how to ensure they are all given an appropriate amount of influence, and how to prevent a bias towards impacts with immediately apparent issues relative to impacts that create issues over a longer time frame.

The fact that change is constantly happening has implications for resource use. Unless the use of a resource is damaging our environment (e.g. global warming, deforestation) then use of even a finite resource should be seen as acceptable. This, however, opens a debate at the heart of the Brundtland definition – equity with future generations. It can be argued that as a resource becomes scarce, market forces will stimulate either greater recycling of that resource or research into substitutes, thus modifying the needs of future generations in such a way as to no longer include that particular resource. This would suggest that consuming finite resources falls within the realm of sustainability.

What the system is, what it is trying to do, is as important as the way in which it does it. This is recognised by the Brundtland definition with its emphasis on meeting the needs of humanity. The Bell and Morse approach also focuses on this by involving all the stakeholders in deciding how to move towards sustainability.

The approach taken in this thesis is to consider sustainability as the sum of two goals.

- First, ensuring the planetary system stays within a set of states that allow for human life, through the identification and protection of natural resources critical to our survival.
- Second, moving the system towards states of a higher quality, or states more adapted for human development.

This approach is in agreement with that of Gudmundsson and Hojer (1996). Defining the characteristics of these states then becomes the key question, and the different theories provide suggestions on how to do so. IE suggests closing the material cycle to reduce resource consumption. The Brundtland report suggests a host of elements covering all areas of social, economic and environmental issues. Bell and Morse suggest determining the characteristics through a participative approach with the relevant stakeholders in the system. Fundamentally, these characteristics determine, or even equate to, the indicators or assessment methods selected to measure these systems.

1 .2. Indicators

Sustainability assessment has an extensive literature in which many different types of indices and indicator frameworks are proposed for measuring the sustainability of systems, products or processes. The multitude of assessment frameworks reflects the wide differences in opinion regarding various author's understanding of sustainability and their priorities concerning it, as well as the variety of systems under assessment and the assessment's purpose. An overview of sustainability assessment methodologies by Singh *et al.* (2012) provides some insight into the breadth of frameworks available, although it should be noted that some of the methods discussed merely consider specific issues related to sustainability rather than the entire picture. In broad terms, assessments exist for assessing countries and regions, such as the framework of the UN Commission on Sustainable Development that was a direct result of Agenda 21 (CSD, 2001), cities (van Dijk and Zhang, 2005, Ross and Underwood, 2010), companies (Veleva and Ellenbecker, 2001, Krajnc and Glavič, 2005, GRI, 2006), products and processes (Schmidt, 2007, Singh *et al.* 2007, Khan *et al.,* 2004, Krotschek and Narodoslawsky, 1996, Afgan and Carvalho, 2004, Begić and Afgan, 2007) and transport (see section 1 .4).

The purpose of sustainability assessment is manifold. Veleva and Ellenbecker (2001) suggest a number of roles for indicators charting company performance, such as informing decision-making within the company, promoting organisational learning, allowing comparisons to be made between companies, self-assessment of the company and encouraging stakeholder involvement in sustainability. Schwarz et al. (2002) echo the role of indicators as guiding decision-making. Krajnc and Glavič (2005) emphasise the need to compare companies. Indicators also imply the direction in which sustainability lies.

To a large extent, sustainability assessment has been developed as an aid to implementing sustainability into various systems. As such, a number of criteria are laid out for indicators in literature, which largely run along similar lines (Veleva and Ellenbecker 2001, Schwarz et al 2002, Krotschek and Narodoslawsky 1996, Bell & Morse 1999, Afgan and Carvalho 2004). Dalal-Clayton and Bass (2002) state that a well-designed indicator framework should be systemic, hierarchical, logical and communicable. The only notable contradiction in the literature is whether or not to allow qualitative indicators, or whether they should be solely quantitative. As a general rule, the frameworks that favour a certain portion of qualitative indicators have a stronger interest in social and societal issues. Another important point, raised by Veleva & Ellenbecker (2001), is that indicators should drive the right behaviour – i.e. improving them should lead the system in the desired direction.

A number of broad schools of thought can be identified on how to most effectively assess sustainability. The first school of thought is the “reporting” style of sustainability assessment. This is characterised by a long list of indicators, split into sections, each covering one particular aspect of sustainability. The result is presented in a fashion similar to a financial report. A typical example of this approach is the Global Reporting Initiative framework (GRI 2006) of sustainability reporting. Azapagic (2004) follows the GRI model quite closely. The result is a comprehensive snapshot of the current performance of the system (typically a company) under review. Critics of this approach suggest that the number of indicators makes decision-making and comparison with other systems difficult. This is compounded by the fact that indicators are not normalised to a single unit.

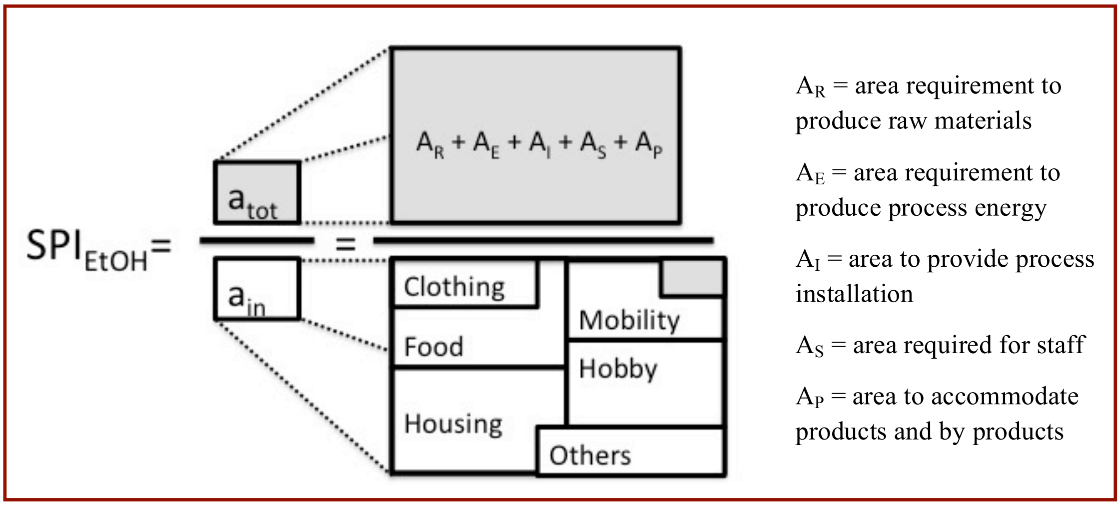

Figure 1 - The SPI is the fraction of two areas: the area a_{tot} needed to embed a process (product or service) sustainably into the ecosphere and the area a_{in} available for one inhabitant. The smaller it is, the easier it is to accommodate that process sustainably – adapted from Krotschek and Narodoslawsky (1996).

At the opposite end of the spectrum lie the aggregated indicators. These combine a number of indicators into a single easily comparable value, commonly termed an index, such as Krajnc & Glavič (2005), Begić and Afgan (2007), Singh *et al.* (2007) and Khan *et al.* (2004). Although a single value indicator appears to make comparisons between different systems very easy and allows immediate evaluation of the progress made towards sustainability, it poses several problems. The biggest issue is that aggregation requires two steps to be taken: normalising the individual indicators to a single unit, and then weighting them. The latter step in particular creates issues, as it introduces additional subjectivity into the assessment and can have a significant influence on the final result.

Aggregated indicators often focus on particular aspects of sustainability to the detriment of others. For instance several authors suggest indices based on the amount of area required to sustain a process sustainably, namely the ecological footprint method suggested by Rees and Wackernagel (1996) and the Sustainable Process Index (SPI) developed by Krotschek and Narodoslawsky (1996). The SPI is an index based on normalising all the relevant factors into a measure of area (Figure 1) Area was chosen as the key dimension because ultimately area determines how much exergy is captured from the sun and hence how much energy is available. The SPI aims to determine the total area required to sustainably provide a given service to a person, and then relate it to the total area available to that person, to give a measure of the relative cost of the service. Whilst these methods offer an interesting insight into the environmental carrying capacity, they don't consider social aspects of sustainability.

The aggregation process necessarily involves losing, or at least obscuring, information. This means that it is not possible to identify why a particular system is performing at its current level, where multiple indicators could help to identify the key areas impacting upon its performance. This can be minimised by preserving the indicators, and making the aggregation process more transparent. A related issue, often raised against aggregation based on a monetary valuation of impacts, such as the index suggested by Pearce and Atkinson (1993), is that of substitution of capital. When aggregating different dimensions of sustainability (environmental, economic, social) into a single value, the implication is that a sufficient improvement in one area can compensate for poor performance in another (for example, an increase in financial capital can make up for a loss in natural capital). Ecosystem experts argue that this is not the case (Rees and Wackernagel, 1999), and that damage to critical ecosystem functions can be irreversible. Finally, aggregated indicators are usually inflexible – they are either tailored to a specific issue or system, or are so general that they don't consider issues specific to certain sectors. Despite all these issues, it should be noted that some composite indices have proven to be useful – for instance the UN Human Development Index (UNDP, 2012).

A number of authors are suggesting a form of 'middle route' in which a small set of core indicators are standardised for all companies, and are supported by a set of supplemental, sector or company specific indicators. Numbers vary for the core indicators. Veleva &

Ellenbecker (2001) suggest using 22 core indicators, mostly fairly simple ones focusing on the main inputs and outputs of the system (resource use, emissions, waste, performance, and training). Martins *et al.* (2007) and Schwarz *et al.* (2002) suggest using far fewer – just 4 or 5. Martins *et al.*(2007) propose the concept of using four 3 dimensional indicators, i.e. indicators that provide information about all 3 aspects of sustainability – social, economic and environmental. As such, they recommend using energy intensity, materials intensity, potential chemical risk and potential environmental risk (social impact is represented by the risk to workers/society).

Several authors place a strong emphasis on the method or framework by which the indicators are generated. Dalal-Clayton and Bass (2002) state that:

> *"Systematic procedures for choosing indicators lay bare the selection and arrangement of issues covered by the assessment and the values involved, and make the construction of indicator-based assessment more transparent than that of accounts or narrative assessments."*

Bell and Morse fall into this category. They have developed a technique called Systemic Sustainability Analysis (1999) that has since been updated to Systemic and Prospective Sustainability Analysis (2003). This is based on a learning cycle which comprises four main areas: (1) getting to know the system and its stakeholders, (2) developing the indicators in a collaborative manner with the stakeholders, (3) plotting them in a graphical representation called an AMOEBA (Figure 2) to allow analysis of trends and finally, (4) examining the process to allow for a new iteration to take place. The method aims to be an ongoing learning process, constantly refining the

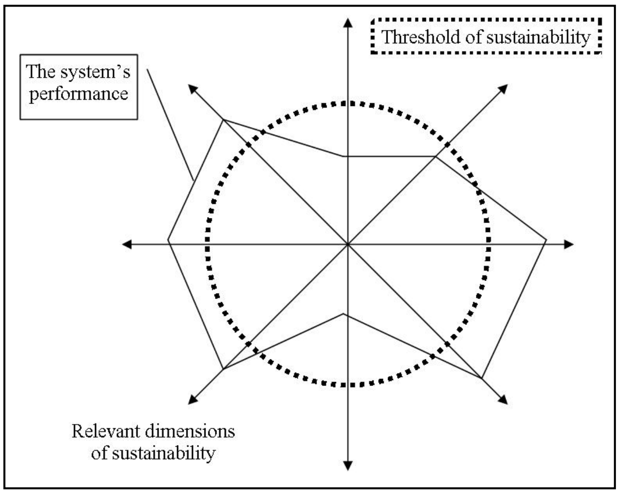

Figure 2 - AMOEBA representation

indicators to suit the needs of the system. Veleva and Ellenbecker (2001) suggest a similar continuous learning loop in their framework. They also share the enthusiasm of Bell and Morse for stakeholder involvement in the selection of indicators. The main difference in their approach is that they define a core set of generally applicable indicators as a starting point. Other authors also point towards participatory techniques and stakeholder involvement to assist in generating indicators, such as Dalal-Clayton and Bass (2002) and Singh *et al.* (2007).

Tahir and Darton (2010) similarly develop a framework for the systematic design of sustainability metrics. Initially applied to the sustainability assessment of businesses (more specifically, palm oil), the method breaks down the process of sustainability assessment into a series of steps. The idea is to reduce the system under review to its component processes and examine the impact of each process on sustainability, according to a previously selected definition. The method is more completely summarised in Chapter 2.

The European Environment Agency developed a general framework for describing environmental impacts, called the DPSIR framework (Driving forces, Pressure, State, Impact, Response - see Figure 3). This corresponds to a systems analysis view of relationships between human systems and the environment (Smeets and Weterings, 1999), where social and economic drivers exert pressures on the environment, changing its state. This change in state creates impacts on human health, ecosystems and materials, potentially eliciting a societal response. Indicators are generated for each aspect. Although there have been some attempts to apply the framework to assessing sustainability more generally (Walmsley, J.J., 2002), there are several criticisms concerning the use of the DPSIR framework for wider sustainability issues. Carr *et al.* (2007) remark that when describing problems on a larger scale, the framework implicitly shifts power to entities able to affects the driving forces (typically national governments, supranational and international organisations) without properly recognising the aggregated impacts of local informal responses. Maxim *et al.* (2009) note that the deterministic 'causal' description of issues does not reflect the uncertainty and complexity of relationships between environmental and socio-economic systems. There is however another issue, which relates more fundamentally to the purpose of sustainability indicators. This purpose is to describe and measure issues pertinent to sustainability – to include indicators describing the causal chain creating the issues is to no longer assess sustainability but to begin to model the system. For example, an increase in car adoption could be seen as a driver for CO_2 emissions – however, were car adoption to decrease, but CO_2 emissions to stay constant (as a result of other interactions within the system), sustainability would not increase, the real issue in this example being CO_2-driven climate change and not car use.

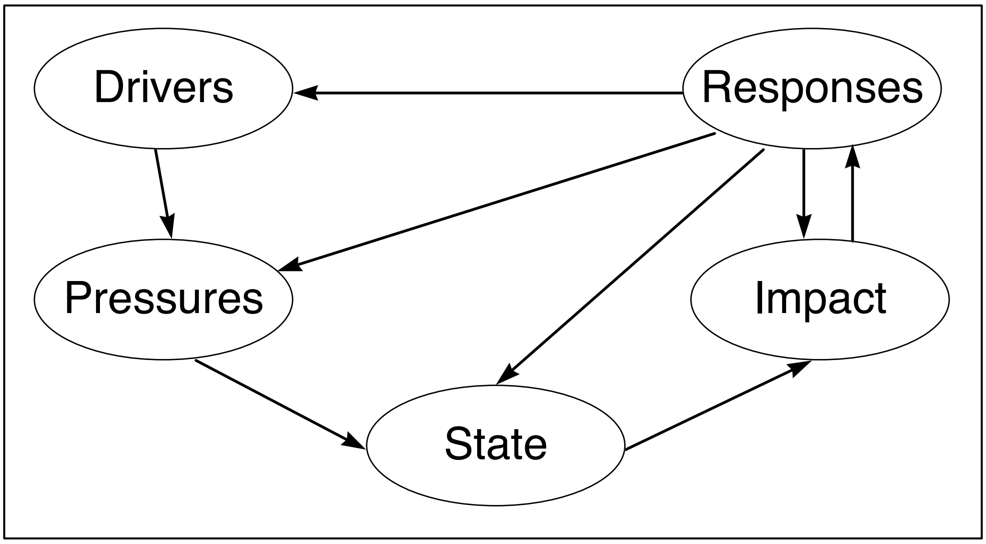

Figure 3 - The DPSIR framework, from Smeets and Weterings, 1999.

An overview of the literature illustrates the wide variety of indicator frameworks available, and the different approaches that can be taken towards sustainability assessment. Reporting style frameworks, whilst exhaustive, are cumbersome and make it difficult to get a sense of overall progress towards sustainability. They lack a driving theme to pull all the indicators together, and it is debatable whether all the indicators are necessary. It seems likely that fewer indicators would be sufficient to drive the system in the right direction. This would however require a greater understanding of the system under review and the issues relevant to it. At the opposite end of the spectrum, composite indicators provide an effective way to summarise overall changes in sustainability, but introduce new issues, including issues related to weighting, substitutability of capital, and loss of information regarding internal system dynamics.

If the goal is to generate a set of indicators that is informative, practical, and an accurate description of the system, then an intermediate way seems most appropriate, with some core indicators supported by system specific supplemental indicators, or possibly some core considerations to include in the assessment such as resource flow. The Veleva & Ellenbecker (2001) method is appealing as it includes a continuous learning loop for perpetually improving the indicator framework. This is also a key feature of Bell and Morse – however, the lack of any core indicators, and complete reliance on participatory techniques whatsoever is a weakness.

Indicators are essentially a means of operationalising sustainability concepts and applying them to a system or process. As such, the indicator set and how the system's sustainability is defined need to be in agreement, and thought must be given to what a sustainable outcome represents for the system under scrutiny. It is interesting that many of the indicator frameworks considered do not address this concern. There are exceptions, such as the UNCSD framework, which clearly links back to Agenda 21 goals (which itself was born out of the conclusions of the Brundtland commission), the Ecological Footprint method (which sets clear thresholds for sustainability) and Bell and Morse, who in their participatory framework include a step identifying what an ideal, sustainable scenario would resemble.

Another issue is the interdependence of indicators. It is assumed that the indicators all vary independently of each other. However, this is unlikely to always be the case. There will for instance be situations where improving on an environmental indicator affects an economic one. This highlights the fact that to be able to truly effect system changes towards sustainability, a greater understanding of system dynamics is required. This would allow indicators to be linked to processes within the system and identify where two indicators might be coupled, giving rise to an emergent property of the system. However this would require accurate, dynamic modelling of the system – simplistic frameworks such as the DPSIR cannot hope to fully capture the complex relationships in socio-economic systems.

1 .3. Tools for Analysis

A number of existing tools can be applied to sustainability assessment, such as life cycle assessment (LCA), which performs an in-depth analysis of a particular product and creates a flow chart (known as an inventory table) of the processes involved in the product system including material and energy relationships (H.A. Udo de Haes 2002). The flow chart includes all the processes involved in creating the product, its use and disposal. These processes are analysed and an impact assessment is made of the extractions and emissions present in the inventory table. Although it focuses mainly on environmental impacts, LCA provides a methodical overview of all the key stages involved in the creation of the product under scrutiny. This helps link impacts with specific processes. LCA was used as the basis for the Life Cycle Index (LInX), a composite indicator developed by Khan *et al.* (2004).

Material Flow Analysis is a method closely linked to Industrial Ecology. Its main aims are the detoxification and the dematerialisation of the ‘industrial metabolism’ (Bringezu and Moriguchi 2002). A set of indicators can be measured charting the flow of materials in and out of a system (for instance a firm, sector or region). The idea is to link the flow of certain materials, products or substances with a particular negative environmental impact and to attempt to reduce that flow as much as possible.

Hierarchical methods are used when dealing with complex multi-criteria problems. Indicator systems describing sustainability fit this profile, and these methods can be useful tools for managing the various interactions between different indicators, as demonstrated by Tahir and Darton (2010), who adopt a method of paired comparisons to assist in weighting different metrics. This method is based on work by Thomas Saaty (1990) on the Analytic Hierarchy Process, a decision support tool. This method is used in a number of other papers, such as Singh *et al.* (2007) and Khan *et al.* (2004).

Exergy is also used in sustainability assessment. The concept of exergy is defined by (Rosen 2002) as:

> *"the maximum amount of work which can be produced by a system or a flow of matter or energy as it comes to equilibrium with a reference environment."*

It does not obey a law of conservation. Rather, it is based on the second law of thermodynamics, and exergy consumption is proportional to the entropy created due to irreversibilities in a process. As such, it allows a distinction to be made between different sources of energy based not only on amount, but also on availability for work. This makes it in many ways a more accurate measure of the value of a resource – energy itself is not valuable, only the work it does has value. This is why exergy is often referred to as a measure of the quality of energy (Dewulf *et al*. 2000). However this isn't unanimous - (Bakshi 2002) argues that exergy is simply a measure of the state of the system and its future ability to do work, not a measure of quality. He cites the fact that a given amount of exergy in electrical form can do more types of work than the equivalent amount of solar exergy as an example of this.

A number of researchers have noted a link between exergy and sustainable development, such as recently Rosen and Dincer (Rosen 2002, Rosen, Dincer *et al.* 2008, Dincer and

Rosen 2006) who suggest that exergy tends to give more meaningful efficiencies than energy. Dewulf *et al*. (2000) suggest a means of assessing the sustainability of technologies and processes based on exergy consumption. Exergy is often used as a means of improving or assessing the efficiency of processes, for instance in the life-cycle of steel (Michaelis and Jackson 1998), or in assessing renewable energy sources (Koroneos, Spachos *et al*. 2003). It has been quite successful in the analysis of single plants and components, but its adoption in the analysis of complex systems has been quite low (Sciubba 2003). Exergy is thus a useful tool if applied to the right problem, but not an all-encompassing tool for sustainability.

1 .4. Sustainable Transport

The objective of this research is to evaluate the sustainability of a transport system through an indicator framework. Previous sections have examined various definitions of what sustainability is as well as different methods by which sustainability indicators have been generated in the past. The literature contains a number of attempts to relate sustainability indicators and transport systems. Some focus more on moving transport towards sustainability, whereas others focus more on assessing the impact of transport.

Defining sustainable transport is the first step towards assessing it. Many papers take the Brundtland definition of sustainable development as a starting point (Richardson 2005, Rassafi 2005, Roth & Kåberger 2002), although it is applied in different ways. Using this definition, a sustainable transport system could be evaluated in terms of how it contributes to sustainable development, i.e. how it contributes to the needs of the present, and how it is potentially contributing to preventing future generations from meeting their own needs.

The EU Council's European Sustainable Development Strategy 2001 defined a sustainable transport system as one that:

> - *"Allows the basic access and development of individuals, companies ands societies to be met safely and in a manner consistent with human and ecosystem health, and promotes equity within and between successive generations.*
> - *Is affordable, operates fairly and efficiently, offers a choice of transport mode, and supports a competitive economy as well as balanced regional development*
> - *Limits emissions and waste within the planet's ability to absorb them, uses renewable resources at or below their rates of generation, and uses non-renewable resources at or below the rates of development of renewable substitutes while minimising the impact on the use of land and the generation of noise."*

This definition considers both aspects of the Brundtland definition, in that it considers present needs (for instance access and affordability) and future needs (for instance resource use, emissions and intergenerational equity). However, it is unfocused and muddled, and reads more like a wish list of various sustainability concepts and transport policy goals than a coherent definition. Some points seem particularly out of place, such as "choice of transport mode".

Akinyemi (2002) offers a particularly appropriate definition of a sustainable transport system, concluding that there are three conditions that a sustainable transport needs to satisfy.

> *"[...] a sustainable urban transport system is postulated to be a transport system that:*
>
> - *consumes or requires at any given time period, rate-dependent resources that are not or will not be greater than the resources that can be internally generated during the time period*
> - *consumes stock-dependent resources less than the resource consumption capacity*

- *produces at any given time period, environmental impacts that are not or will not be higher than the environmental capacity"*

This definition addresses the latter half of the Brundtland definition – i.e. ensuring that future generations can meet their needs, and that the transport system does not prevent them from doing so. Akinyemi presents two further definitions: that of a *sustainable and developing* urban transport system, and that of a *sustainable and developed* urban transport system. A *developing* system meets the aforementioned conditions, but also provides "*developing levels of mobility, accessibility, and safety of movement for all socioeconomic groups at any given time period*". A *developed* system has met the ideal levels of these criteria. These definitions incorporate a measure of the ability of the transport system to meet present needs. However, the social aspects are limited to accessibility, safety and mobility, qualified by a consideration of equity.

1 .4.1. Tools for Change

Many sustainable transport papers focus on how to change the transport system to make it more sustainable, albeit using a variety of tools and with different specific goals. This is clearly an important task; however most of these papers lack an in depth analysis of what constitutes a sustainable transport system, or indeed any systematic method of generating the indicators they use. As a result, several papers have a narrow focus when it comes to indicators, emphasising a few key issues related to sustainability but not fully addressing it.

For instance, Roth and Kåberger (2002) concentrate on the reduction of CO_2, highlighting issues in current business sustainability accounting. Similarly, Grimes-Casey *et al.* (2009) suggest using CO_2 as a quantitative target to drive 'sustainable mobility'. Zachariadis (2005) proposes a transport simulation and model for assessing policies, using emissions, noise, congestion, accidents and energy intensity as indicators. Richardson (2005) develops a framework for analysing the factors contributing to five main transport issues: safety, congestions, fuel consumption, vehicle emissions and access. Ford (Schmidt, 2007) has produced a Product Sustainability Index, which examines the life cycle impacts of their vehicles. It is a product centric approach, focusing more on reducing the impact of the vehicle than on the performance of the transport system in which it resides. These approaches are narrow; on the other hand, they provide tools to bring about positive change.

1 .4.2. *Assessing Sustainability*

Other papers are more focused on a general assessment of the sustainability of transport systems. These papers generate a broader range of indicators covering the three sustainability domains: social, economic and environmental. Examples of this include Kennedy (2002), who performs a sustainability assessment for transport within a city region, and Nicolas *et al.* (2003) who propose a set of indicators for evaluating the impacts of mobility within a city. In each case, the indicators chosen are sensible. However the studies lack a framework by which to abide or a system for the selection of indicators, which appear to be chosen arbitrarily. Fedra (2011) similarly bases his indicator set on an amalgamation of UNCED, UNEP and EEA indicators, with no real explanation as to how they were selected. Special attention ought to be paid to the mobility indicator set produced by the World Business Council on Sustainable Development (WBCSD, 2004). Although no real framework or method is displayed for selecting the indicators (aside from mentioning extensive stakeholder consultation), the indicators produced cover the key issues. The study assesses possible trends of some of these indicators into the future, and sets goals based on the most important issues.

Gudmundsson (2003) lays down criteria for indicators for sustainable mobility, which should:

- *"provide a comprehensive picture of the current environmental pressures and impacts from mobility, as well as developments over time;*
- *Identify the causal factors (technological, social-economic, etc.) behind these developments and describe their relative contribution to changes in each environmental impact;*
- *Link environmental pressure indicators to relevant sustainability reference values and targets to indicate gaps and gauge progress;*
- *Measure the fulfilment of policy commitments, objectives, and targets for sustainable mobility;*
- *Feed information from the first four points back into the processes where mobility-relevant policies are formulated and implemented."*

Aside from being strongly focused on environmental aspects, with little emphasis on social or economic concerns, these criteria are interesting in that they attempt to link the indicators with environmental targets and policy goals. They also suggest performing an analysis of the system to determine the causal factors behind changes in environmental impact, presumably as means of informing policy decisions. The issue becomes how to set reference values and targets as well as the policy objectives and targets. It implies that government needs play a central role in sustainability assessment. Regardless, the idea of identifying causal factors driving changes in the system is a useful one, as in order to implement positive changes in a system these need to be understood. These causal factors need to be differentiated from sustainability indicators, however. Behrends et al. (2008) suggest that "*indicators do not necessarily need to quantify the impacts which are the last elements in the causal chain*". However, it can be argued that the indicators should focus on describing the issues affecting stakeholders (and hence sustainability) wherever possible, and that to describe elements further up the causal chain is to begin trying to solve the problem rather than describe it. This is a similar criticism to that of the DPSIR framework in section 1 .2.

Black (2002) proposes a unique framework of performance indicators and analytical methods for assessing the sustainability of an urban transport system, again focusing on the three domains, considering efficiency and equity. Black has noticed that much of the literature “fails to link indicators with higher –level goals for the system” and attempts to remedy this. The method suggested is based on a hierarchy, whereby the broadest goal is placed at the top of the tree, and subsequent levels outline progressively more precise objectives. Each sub-objective is associated with a measurable attribute (which acts as a target) and a suggested “policy instrument” with which to achieve it. This method identifies the problems and provides direction and a goal to work towards, ultimately driving the system under scrutiny towards sustainability. However, it only provides an incomplete assessment of the sustainability of a system, in that it fails to account for the impacts of the service, i.e. the mobility provided.

Another unique framework is a product of the European Commission SUMMA (SUstainable Mobility, policy Measures and Assessment) study (Walker et al., 2006). A systems approach is adopted, linking the EU Council’s definition of a sustainable transport system to a series of environmental, economic and social goals, and noting that “the transport system is not an end in itself, but rather a means to other ends” and that the aim should be to “make sure the outputs from the system contribute to the sustainable development of society”. The indicators describe a series of ‘outcomes of interest’ which affect various stakeholders in the system. Despite this, it is not always clear why some indicators are included, and what the direction should be. The overall focus is on the complete transport system, and not on any particular mode. Equally, the indicators are not actually applied to any real world case.

1 .4.3. *Thermodynamic assessment*

A number of authors have applied embodied energy based techniques for assessing urban systems (Treloar *et al*. 2001, Troy *et al*. 2003, Huang 2003). These consider the embodied energy in cities, and hence include some assessment of transport costs. However, these are not particularly in-depth assessments, as the main focus of these papers is not transport. Huang *et al*. eschew any consideration of vehicles, and focus solely on the supporting infrastructure (mainly roads).

However, there are examples of these techniques being applied more specifically to transport systems. Federci *et al.* (2008, 2003) conduct a thorough thermodynamic study of a variety transport systems, in order to compare different modes and improve them. They apply a range of methods, namely embodied energy analysis, exergy analysis and emergy analysis. The results of the studies highlight first the importance of indirect energy and material usage (either embodied or related to infrastructure) on the overall energy and material intensities. The authors note however that exergy is better suited to the analysis of individual processes; this is due to it being impossible to assign an exergy content to the to the passenger-km or ton-km which comprise the system's product. This greatly diminishes the value of exergy as a tool in sustainable transport assessment. However, they introduce a useful indicator using exergy, a ratio of exergy use per p-km of the best performing vehicle on the market working at maximum capacity and average exergy consumed per p-km in the transport system (excluding embodied exergy). This gives a measure of the additional exergy loss caused by the transport system itself.

These types of studies approach the sustainability question from an engineering based perspective. They are not true sustainability assessments in that they do not address the various sustainability issues (environment, economy and social). Nevertheless the application of thermodynamic methods is interesting and highlights their potential as a tool for assessing transport systems.

1 .5. Service Quality

The area of transport assessment that is most difficult to measure is the contribution of transport to sustainable development. It is relatively easy to identify the impacts of a transport system when considering it as a product; it is the same as any other product one might wish to assess. A variety of sustainability frameworks can be used, supported by life-cycle assessment. However, this does not provide a complete picture of the transport system's contribution to sustainable development. Indeed, there needs to be some recognition of why the system exists in the first place, i.e. to provide transport. To do this requires the transport system to be evaluated as a service provider, and transport itself as a service. Ferreira (2007) identifies four main types of mobility benefits: economic (access to jobs, education, services), personal (career, social life), equity (poor mobility compounds social inequality) and increased travel options in case of emergencies.

Evaluating the impacts of these mobility benefits is difficult, as they depend strongly on the specific use to which transport is put. Evaluating this is both difficult and highly subjective. It also becomes difficult to separate the contributions of the environment on how transport is used, and the effect of transport on its environment. This is highlighted by Kennedy (2002), who notes that transportation can be a driver to the economy but that it is difficult to quantify this impact because it is "*hard to separate the individual contributions of capital, labour and infrastructure*" to the economy. There are many examples of this issue. For instance, out of town retail parks are made possible by transport, but they also act as a driver of transport. A final issue is that almost any trip can be accomplished by a variety of modes. This makes it difficult to assess the unique impact of any individual transport system.

However, assessing the service provider (i.e. the transport system) based on the quality of service it provides is a more feasible objective. Indeed, an example of this is provided by Kennedy (Kennedy 2002), who includes a measure of 'level of service' in his sustainability assessment. It is described as a combination of "*comfort, quality of mobility in terms of access and speed and satisfaction of other passenger demands*", but the main measure used in the study is average speed for a trip.

Services differ from products in a number of ways. Parasuraman *et. al* (1985) note that services are "*performances rather than objects*", and hence cannot be tested, measured or verified ahead of time. They also state that services will vary "*from day to day, producer to producer and customer to customer*" and that production and consumption of services are interrelated in that consumer input is often critical to the service quality. These three characteristics are remarkably pertinent when applied to a transport system, even to cars. Although a car can be tested ahead of time, the trips taken in it will vary based on a number of outside influences, and cannot be predicted that accurately. As such cars are designed to handle many different types of trips or situation. And the consumer himself contributes to the overall service quality, for example through his driving. Discerning the key components of service quality, especially with respect to mobility, is an important first step towards designing a complete evaluation.

Parasuraman *et al.* identify a number of 'determinants of service quality': reliability, responsiveness, competence, access, courtesy, communication, credibility, security, understanding, knowing the customer and tangibles ("physical evidence of the service"). Not all of these factors are universally applicable to transport systems. For instance responsiveness, courtesy, credibility, communication, don't work when applied to cars – this is due to the fact that cars are essentially a 'self-service'. This is one of the key differences between cars and other modes of transport, and one of the points that cause comparison with other modes to be difficult.

One of the relevant factors is reliability. This is emphasised in a number of other studies, such as (Bell 2000, Bates *et al.* 2001, König and Axhausen 2002) which all suggest that reliability is a strong concern when dealing with transport. Waiting times are actually less of a problem than uncertainty.

Other factors having an influence on service quality highlighted in transport literature are safety, security, degree of flexibility and accessibility (with hourly opportunities to travel as a potential indicator). Attractive values of cars are listed as comfort, speed, convenience and individual freedom. For public transport, frequency, comfort, travel time and fare level are highly valued (Beirão and Sarsfield Cabral 2007).

An important aspect of sustainability is equity, and in transport this includes addressing the mobility needs of different socio-economic or demographic groups of special concern. These include the elderly community, but also youth, disabled, families, the unemployed and the poor. An important aspect of evaluating a transport service would therefore be including an indicator, or indicators, that address how well the service addresses the needs of specific groups that make up a community, especially those groups that are often marginalised (Ferreira 2007).

Beirao and Sarsfield Cabral (2007) note that "travel behaviour is influenced by the service level of the transport system." Fujii and Kitamura (2003) note that this influence is affected by psychological factors. This suggests that an easily understandable measure of service quality could be useful in helping people to change their perceptions, outlook and habits. However, measuring service quality creates some difficulties. Beirao's research (a survey of user attitudes) implies the problem is twofold:

- People have different priorities. Therefore producing a measure of 'objective service quality' becomes very difficult, if not impossible.
- These priorities change depending on the purpose of the trip. For instance, boot space in a car might be unimportant when driving to work, but become an issue when going shopping. On the other hand, reliability becomes much more important.

A different perspective on the issue is provided by Winston (1985), who states that:

> *"The various components of service quality – travel time, comfort, reliability, etc – are recognised as important attributes of a transportation mode. The users' valuation of these attributes will depend on the characteristics of their utility function, itself dependent on their tastes and the activity to be performed at the destination."*

A utility function is a function that assigns worth to different outcomes according to the stated goals of a decision maker. As such it is a reflection of his preferences. (Game Theory, Encyclopedia Britannica)

Even if the utility function of the user is not known, it is possible to generate a list of the various components of service quality and assign a value to them. This would leave the user free to apply his (or her) own utility function to the problem, but with better, more in depth and accurate data on which to base his decision.

1 .6. Conclusions

A survey of the literature suggests that there are some gaps in sustainability assessment, specifically in the sustainability assessment of transport. The main weaknesses of current assessments are:

- An incomplete understanding of the concept of sustainable development, especially in terms of its application to transport, leading to the omission of key indicators

- Many indicator sets lack a clear framework or methodology by which the indicators are selected. This often translates to a lack of a driving theme, with no direction or reasoning for the indicators.
- Those indicator sets with a stronger theoretical basis tend to lack any practical applications, at least with respect to transport systems

The method suggested by Darton and Tahir (2010) appears to provide a strong starting point for generating a sustainability assessment of transport systems. Although it was designed with the assessment of manufacturing businesses in mind, it is rigorous, consistent and transparent, with a clear reasoning behind the choice of each indicator. It is interesting to widen its application to a service based system.

A number of tools should be considered in addition to the method, such as life cycle assessment and embodied energy. Furthermore, the product of the system (in the case of transport, the mobility provided) must be considered in terms of its contribution to sustainable development.

Chapter 2

Applying the Process Analysis Method to Transport

The Process Analysis Method (Tahir & Darton, 2010) is a way of analysing a system to generate indicators that describe its impact on sustainability. It was applied to a transport system, more specifically the car fleet, as a means of testing its wider applicability and to construct an indicator set describing the sustainability of a transport system.

2 .1. The Methodology

The method (Figure 4) breaks the system into individual processes, and sets a definition of sustainability and a system boundary. It then evaluates the impact of each process within the boundary on the three sustainability domains: environmental, economic, and social. Each impact causes one or more issues, affecting one or more stakeholders, known as external impact receivers (EIR). One or more indicators are then selected to describe these issues. By looking at the processes individually, it is possible to identify all the ways they can interact with their environment, i.e. all the possible impacts. A working definition of sustainability is used to ascertain whether the impacts are causing issues for particular stakeholders (either positive or negative). If the impacts are not affecting any stakeholders they are not relevant. Since the issues are checked against the chosen definition of sustainability, the method produces consistent results for a given definition. Furthermore, each indicator can be linked to a specific issue, affecting a specific stakeholder group, and each issue can be linked to a specific impact. In this way PAM produces transparent results, as it is possible to trace indicators back to a specific process. Internal Impact

Generators (IIG's) are identified as decisions within the system, which influence its operation.

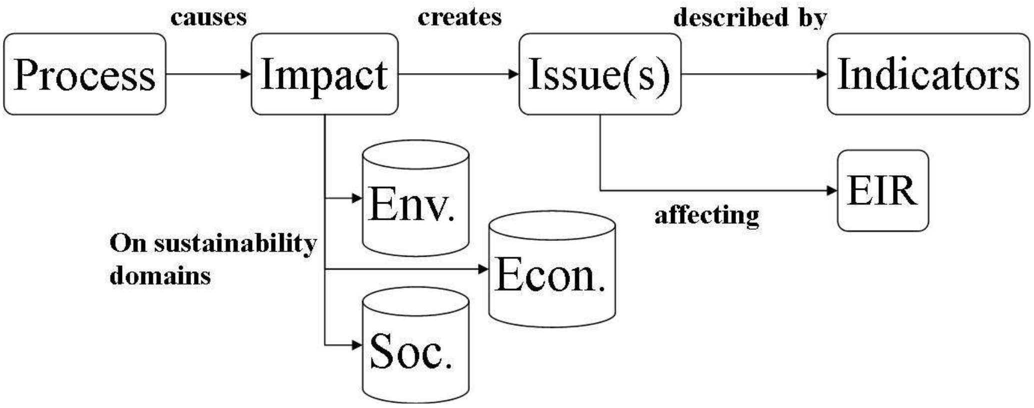

Figure 4 - Schematic of the PAM procedure.

2 .1.1. *Overview of the System*

An in-depth overview of the system (or business) identifies the major processes, along with the associated input/output and stakeholder interests. The main processes involved in a car-based transport system are shown in Figure 5. These loosely follow a car life cycle. A process can have an impact either through the resources it uses (inputs), or the resulting products and waste (outputs). By examining the processes in Figure 5, it is clear that the

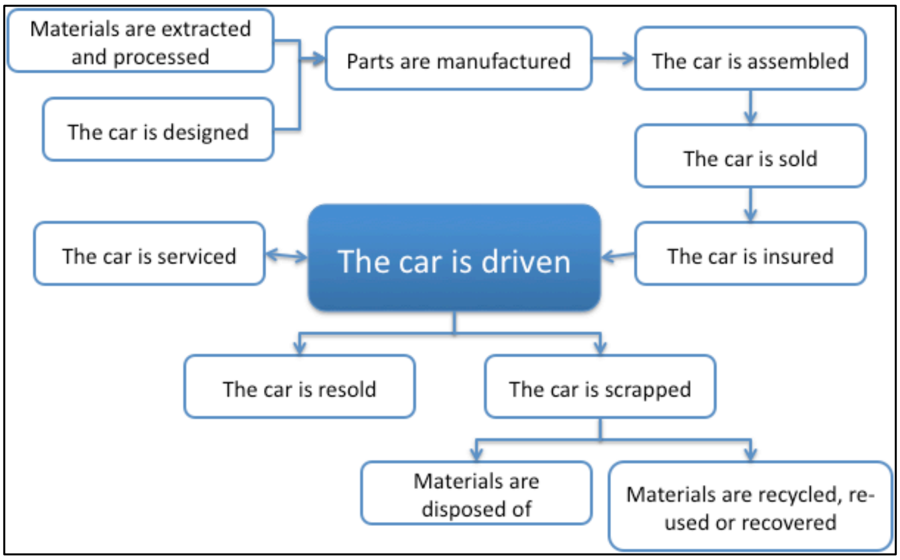

Figure 5 - Overview of processes in car-based transport system.

ultimate product is mobility. The resources consumed by the system were split into five broad categories: energy, materials, land, water and workforce. Once the analysis is completed, the system can be reduced to a single, top-level process, 'providing mobility', with inputs as the sum of all resources used and outputs as mobility, and the sum of all wastes produced. The result, shown in Figure 6, is analogous to a heat engine. A benefit of this approach is that it reduces the problem to one pertinent to any transport system – all transport systems consume resources, and provide mobility. Therefore the indicators produced should be broadly applicable to all transport systems.

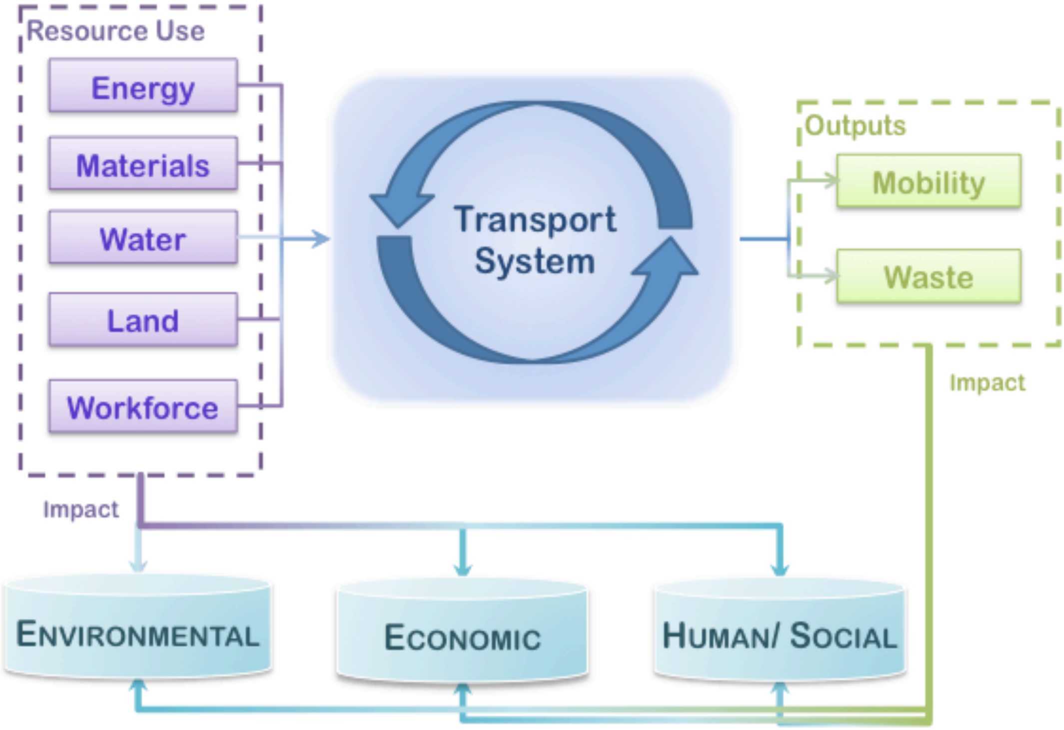

Figure 6 - Top-level view of transport system sustainability impacts.

2 .1.2. *Working Definition of Sustainability*

The working definition of sustainability informs the criteria by which the system will be evaluated. Akinyemi's transport-related definition is the best starting point, due largely to its recognition of the role of a transport system.

It is modified in a number of ways. First, elements of the Rennings and Wiggerings rules for sustainable resource use (Rennings and Wiggering, 1997) are included, as they deal with the use of stock-dependent or non-renewable resources in a more useful manner. A further distinction is made relating to stock-dependent resources: distinguishing between recoverable and non-recoverable resources. 'Recoverable resources' refers mainly to abiotic resources such as steel which are not actually consumed in such a way as to render them useless. Non-recoverable includes all resources that are effectively degraded once consumed, such as fossil fuels, plastics or fissile materials. In a sustainable system, all recoverable materials should be recycled or re-used. Accessibility is removed as a criterion for sustainable transport – the transport system is deemed to be a provider of mobility – accessibility reaches into the realm of how the mobility is used. The clause on environmental impacts is curtailed to cover only environmental aspects, removing the consideration of 'human, physical and socioeconomic conditions'. Instead, a new clause on social impacts is added, which we suggest deals with these aspects more effectively.

Hence, a sustainable transport system:

- delivers for all socioeconomic groups and at any given time period, levels of mobility and safety of movement that are equal to what the society considers to be ideal levels.

- consumes or requires at any given time period, rate-dependent resources (e.g. finance) that are not or will not be greater than the resources that can be internally generated during the time period,
- consumes non-recoverable stock-dependent resources in a quasi-sustainable manner by limiting their rate of depletion to the rate of creating or developing rate-dependent substitutes,
- recycles recoverable stock-dependent resources insofar as this is necessary,
- produces at any given time period, environmental impacts that are not or will not be higher than the environmental capacity,
- in its provision of mobility, maximises positive impacts on society (e.g. employment), whilst minimising undesirable human/social impacts (e.g. negative health effects).

This definition echoes the Brundtland definition in that it considers the needs of the present (i.e. mobility) whilst ensuring that future generations are safeguarded against environmental impacts and resource depletion.

A practical issue with this definition, which is present in most definitions of sustainability applied to specific systems, is the difficulty in allocating what constitutes acceptable levels of resource use and environmental impact for the system in question relative to other systems. For example, how much of a society's acceptable levels of carbon dioxide production (i.e. within the environmental capacity) should be allocated to transport? Another issue is how to balance improvements in one area against deterioration in another. For example, an increase in vehicle cost versus a decrease in emissions. These are societal problems – the answer depends on what the society in question prioritises. Tahir and Darton (2010) derive two business perspectives from the Brundtland definition which help to address these issues by providing a basis on which to assess impacts and generate indicators that will push the system towards sustainability. These perspectives are:

- **Resource efficiency** [which] measures how effectively the capital is used or created (change can occur in both the amount of capital, and its quality); and
- **Fairness in (dis)benefit** [which] means, with regard to using or changing the capital, both (a) how fairly the benefit is distributed, and (b) how fairly disbenefits are distributed

These perspectives are not criteria for sustainability, but by maximizing them the system should become more sustainable.

This definition highlights an important point related to the scope of the study. No attempt is made to place a value on the benefits (or disbenefits) of the purpose attributed to the trips performed by users of the transport system. Whether the mobility produced is used for shopping, work or leisure purposes is irrelevant to how it is valued. Mobility is considered to be a positive, in that it is the product of the system under consideration. The study is concerned with how efficiently it is produced, how well it is provided (quality) and how equitably it is made available to society; in other words, how well it meets the needs of society. As a consequence, impacts such as the benefit of better transport links are not considered. If better transport links were created the benefit would be reflected by an increase in the quality and/or quantity of the resulting mobility, or in the efficiency of its provision. The view that mobility is an unqualified positive is not universal: Banister (2008, 2011), Lucas et al. (2007), and the SUMMA study (Walker et al, 2006) all point towards the importance of accessibility (to jobs, goods and services). This is discussed further in 2 .5.

2 .1.3. *Identify the System Boundaries*

A spatial and temporal system boundary must be set. The spatial boundary is the physical size of the system boundary and the temporal boundary defines the period over which impacts are considered. In this study, the system under consideration is the UK car fleet, and the processes that support it. The boundaries are chosen to be in line with the goals of the study, the main goal being to measure the impact that the system is having on the sustainable development of our society, and our environment.

The spatial boundary is the geographical boundary of the UK - all processes of the system occurring within this boundary are assessed in terms of their impact on sustainability. Hence we include impacts related to the manufacture, sale, operation, insurance, maintenance and disposal of the car. However, some processes such as manufacture involve a long supply chain, which reaches well beyond this boundary. Ignoring the impact of these supply chains could easily misrepresent the overall sustainability of the system. However, pragmatically a line does need to be drawn. It was deemed appropriate to include data on impacts occurring outside this geographical boundary when these were significant relative to those occurring within. An example would be the embodied energy and emissions in the materials used in vehicles, or the impact of imported vehicles.

One aspect, which is excluded from our study, is the impact of the supporting infrastructure to the car fleet such as roads, refuelling stations and miscellaneous buildings involved in the aforementioned processes. In the case of roads, this is chiefly because is difficult to discern how much should be attributed to the car fleet. The road network benefits a number of different modes of transport, such as bicycles, motorcycles, buses and HGVs. The costs could be split according to usage - the proportion of traffic represented by the car fleet. However, if cars didn't exist, most of the road network would probably still have to be in place; it is unreasonable to assume that the infrastructure impact of the road network would go down linearly with a reduction in traffic. Figure 7 shows how growth rates for the vehicle fleet are consistently 8-12 times higher than growth rates in the road network. A similar point applies to the refuelling infrastructure. In terms of the various buildings involved in supporting the car fleet (factories, offices etc.) the issue is simply that of data availability.

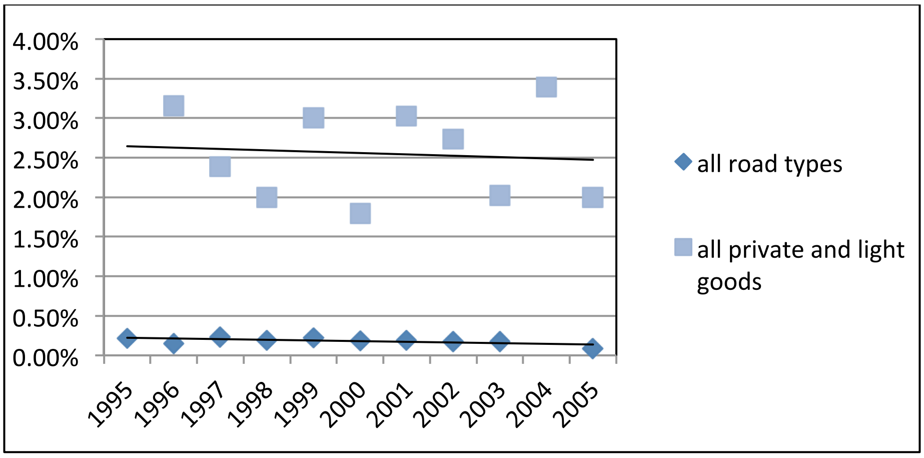

Figure 7 - Growth rates of car fleet (number of vehicles) and road length (km).

Impacts can occur on different scales, in that some are extremely localised (congestion at a particular junction), whilst others are global (climate change). Given that the study is done on a national basis, local impacts were considered insofar as they were statistically significant on a national scale. An effort was made not to overly aggregate data.

2 .1.4. *Identify the Domains and Capital of Sustainability*

The method identifies three sustainability domains on which the system processes can have an impact. These are the economic, environmental and human/social domains. These domains are considered to contain stores of value: financial capital, natural capital and human/social capital respectively.

Natural capital refers to the store of value contained in the natural world. Financial capital includes money and financial assets, as well as tangible and intangible assets that could be turned into cash. Human and social capital is the value contained in human beings – both the intrinsic value of a person and the extra value people acquire by being part of a group with interpersonal interactions. The system processes affect these stores of value, contributing to them, preserving them or depleting them.

2 .1.5. Identify the Internal Impact Generators

The ‘Internal Impact Generators’ (IIGs) are the decisions taken, or policies implemented by stakeholders within the system that influence the system’s impacts.

For the car fleet, four IIG’s were identified.

User decisions: Users exert an influence through the choices they make, in terms of mode, amount of travel, type of vehicle purchased/rented, driving style etc. They are also impact receivers – they are affected by the consequences of their own resource use (although many of these consequences are not solely borne by them). There is therefore an element of feedback in the system, as users modify their decisions according to the consequences of previous choices. Users value the various attributes of a transportation system according their own tastes and the activity they intend to perform at their destination. The users are not a homogenous group or entity, and the impact they have is an emergent property of a series of individual choices. This changes the way they react to feedback.

Government policy and regulation: This includes the standards (such as emissions standards) that vehicle manufacturers need to abide by, management of the car fleet through vehicle licensing and MOT, management of the road network, and taxation on fuel and cars, as well as standards and legislation for disposal. Finally, the energy policy affects the impacts of energy use.

C*ar manufacturer design, research and development choices*. Whilst it may at first appear that the car manufacturers are purely impact receivers, in that they respond to consumer demand and government regulation, they play a key role in choosing what technologies to research, develop, and implement in their designs. Technology choice has a crucial impact on the system constraints, as it influences the energy sources used, the material use, the land use, as well as other factors such as energy efficiency.

C*ar manufacturer sustainability policy and practices.* Although sustainability policy could also be considered a response to consumer demand and government regulation, the extent of car manufacturer's commitment to environmentally and socially sustainable practices will also influence the overall impact of car production and operation.

2 .1.6. Identify External Impact Receivers and Issues

The changes to the stores of value, caused by the activities of the Internal Impact Generators, will affect the owners or guardians of the capital. These owners are referred to as External Impact Receivers and are composed of the various stakeholders in the system. Their concerns regarding a particular impact constitute 'issues'. Impact Receivers are identified in order to ascertain the relevance of a particular impact. If no Impact Receiver can be identified then the impact does not warrant further attention.

Five EIRs were identified in this study.

Communities represent the collective sum of people that form human societies. In this instance they consist of the UK population between 1995 and 2005. They are affected by a wealth of impacts, including those that affect the environment or biodiversity in so far as impacting these will have a 'knock-on' impact on human societies, for instance in terms of health effects. Communities are usually represented by other groups such as government, but also NGO's.

Future communities represent the communities that will exist at some point in the future. They are largely affected by the same issues as communities, but over a longer time frame.

They are also affected by resource depletion. Again, these are represented by government and NGO's, which act on their behalf.

The *users* represent the users of the UK car fleet in 1995 and 2005, namely passengers and drivers. They are mainly affected by service quality, but also by cost. Users are a subset of communities, and are affected by all community impacts as well as user specific impacts.

Employees represent the collective sum of people employed in the motor industry and associated services in 1995 and 2005. They are also a subset of communities, but are affected by virtue of their employment in the motor and/or service industries.

Capital providers represent the collective group of investors who benefit directly from the profitability of the motor industry and associated services in the UK.

In the final step, indicators are selected to describe the issues; metrics are developed for each indicator to measure the severity of the impact caused by the Internal Impact Generators on each store of value – environmental, economic and social. These are presented in the following section.

2 .2. Environmental Indicators for Transport System Sustainability

Environmental impacts occur as a result energy, material, land and water consumption and resulting waste emissions. From the above analysis processes were identified with impacts relevant to the environmental store of value (see Figure 8). These can be divided broadly into manufacture, operation, and disposal of the car fleet.

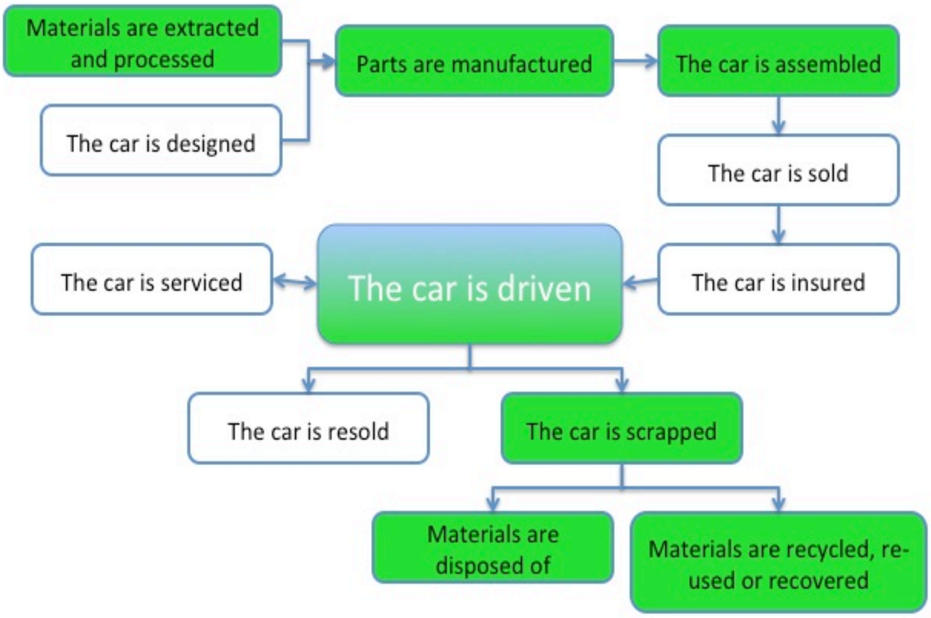

Figure 8 - Processes with environmental impacts.

The issues at stake fall into two main categories: environmental damage caused by waste (emissions to air, water and ground), and the depletion of natural resources. The environment is a common good, damage to which ultimately affects both current and future communities, through impacts to health, crops, building materials, climate, as well as natural resources (vegetation, biodiversity) on which the ecosystem (and hence humanity) depends. There is a rich literature from which to draw information on these types of environmental impacts. In particular, the literature surrounding Life Cycle Assessment provides detailed lists of pollutants and their respective impacts, including weighting factors based on their potency (Baumann & Tillmann, 2004, IChemE, 2004, Guinee *et al*., 2002 and AEA, 2010b). Natural resource depletion affects future communities – although there is uncertainty surrounding the exact resource needs of future communities, it is clear from current trends that energy, materials, water and land will be in even greater demand. The full list of indicators is presented in Table 1. The impacts of each process have been aggregated according to the top-level inputs and outputs identified in Figure 6.

Input/ Output	Impact	Issue	EIR	Indicator	Metric
Energy	Acidification	Increasing acidity in soil, lakes & rivers	Communities	Emissions of pollutants characterised in Guinee *et. al* (2002)	Te/y SO_2 equivalent
	Eutrophication	Ecosystem degradation	Communities	See above.	Te/y PO_4 equivalent
	Global Warming	Stores heat, causing global warming	Future communities	See above.	Te/y CO_2 equivalent
	Photochemical Oxidation	Ozone causes irritation to respiratory systems and can damage vegetation and materials	Communities	See above.	Te/y ethylene equivalent
	Ozone Layer Depletion	Depletion of stratospheric ozone causes greater incidence of UV	Communities	See above.	TE/y CFC-11 equivalent
	Ecotoxicity	Freshwater toxicity, marine toxicity, terrestrial toxicity from emissions to air	Communities	See above.	Te/y 1-4 DCB equivalent
	Particles	Degrades materials	Communities	PM_{10}	Te/y
	Primary energy resource depletion*	Fewer available resources to meet future needs	Future communities	Total non-renewable energy consumed	GWh/y
Material Use	Waste to landfill	Use of land for waste disposal, emissions	Communities	Total waste	Tonnes/y
	Abiotic resource depletion*	Use of non-renewable, non-recyclable and non-reusable materials leave fewer resources available to meet future needs	Future communities	Net materials lost	Te/y Antimony equivalent
	Material processing and extraction	Release of polluting substances	Communities	Heavy metals, toxic chemicals, solvents released	Tonnes
Water	Contamination of water	Treating water involves an energy cost, dirty water can spread contaminants and bacteria	Communities	Quantity of water used	litres
*includes renewable resources used at an unsustainable rate					

Table 1 - Environmental indicators for transport system sustainability.

2 .3. Economic Indicators for Transport System Sustainability

Economic impacts take the form of either costs or financial gains affecting External Impact Receivers. Every EIR is impacted financially in some way, be they *capital providers, employees, users, communities* and *future communities. Capital providers* are affected most directly by economic impacts, given that their stake in the system is primarily a financial one. Here the important factor is the value added by the various industries related to the car fleet and its component processes. The *community* perspective accounts for the various economic costs incurred by impacts on health and the environment. It also considers the costs incurred by government in managing the transport system, although these largely concern infrastructure, which is outside the system boundary. The *community* also benefits (indirectly) from the taxes collected by government. The *users* are affected insofar as they experience direct benefit from the mobility provided, although attributing a direct economic benefit is difficult, as not all trips provide one.

The difficulty with assessing economic impacts is the fact that most impacts in either the environmental or human/social domains can also be described in monetary terms. For instance *employees* are rewarded by the payment of salaries and other work-related benefits. This could be an economic impact, but when considering the *issue,* namely that employees are able to support themselves and their families, it could just as well be described as a human/social impact, given that the issue is largely related to human welfare. Similarly, *users* incur costs linked to car use, such as maintenance, insurance, operating costs and depreciation, but when looking at the *issues* raised by those costs, the main one is affordability – again an issue related to community welfare.

It is also worth noting only impacts on financial capital are considered – for instance, in the case of accidents, the impacts included are lost economic output, medical and healthcare costs, material damage, police and fire service costs, insurance administration and legal and court costs. Impacts of pain, grief and suffering are not included despite monetary cost estimates being available (these are based on *willingness to pay*). These impacts are deemed to affect the human/social store of value, and are therefore accounted for in non-monetary terms. Similarly, impacts of emissions on health are only included insofar as they affect economic output and medical and healthcare costs.

2 .3.1. Impact of mobility on the economy

Mobility impacts the economy, and thus the *community,* in a number of ways. The Eddington Transport Study (DfT, 2006c) identifies seven drivers through which transport can impact the economy:

- Increasing business efficiency through time savings and improved reliability […].
- Increasing business investment and innovation by supporting economies of scale or new ways of working. […]
- Supporting clusters and agglomerations of economic activity. Transport improvements can expand labour market catchments, improve job matching and facilitate business to business interactions. […]
- Improving the efficient functioning of labour markets, increasing labour market flexibility and the accessibility of jobs. […]
- Increasing competition by opening up access to new markets. […]
- Increasing domestic and international trade by reducing the costs of trading. […]
- Attracting globally mobile activity to the UK by providing an attractive business environment and good quality of life.

The report also notes that:

> "*Transport cannot of itself create growth: it is an enabler that can improve productivity when other conditions are right. Economic growth itself causes rising transport demands which, if left unchecked, can put the transport network under strain, damaging productivity and competitiveness.*" (Dft, 2006)

As a result, although transport is clearly a necessary component of our economy, quantifying its economic impact in monetary terms is difficult, and quantifying the impact of a particular mode even more so. Assessing the impact of a specific mode requires an understanding of 'how much better' that mode is for the economy than other modes, which is dependent upon two main characteristics of this mobility: is the journey possible (connectivity) and how long will the journey take (speed)? Should the system come under strain as a result of growth, congestion will occur, causing a drop in speed – as a result, capacity (and limit thereof) is also an important consideration. These characteristics relate to the performance of the system, and are as such captured elsewhere (see 2 .4 Human/Social Indicators for Transport System Sustainability). Nevertheless, some proxy indicators are generated to show the economic importance of the transport system under assessment. The full list of indicators is shown in Table 2.

Input/ Output	Impact	Issue	EIR	Indicator	Metric
Work-force	Value is added through the use of workforce	Generates profit	Capital Providers	Value added by automotive industry	£
Mobility	Accidents/ Casualties	Financial cost of accidents to society	Communities	Cost of physical damage, medical costs, emergency services	£
	Mobility is used to support economic activity	Mobility is used for business	Capital Providers	Business Km/yr	Km/yr
		Mobility is used for commuting	Users	Commuting Km/yr	Km/yr
		Speed of mobility - time is saved relative to other modes	Users	Cost (in terms of value of time) per km	£/km
	Additional revenue collected by the government as a result of car fleet operation	Government collects tax revenue to pay for public services	Communities	Fuel duty and vehicle Excise Duty Collected	£
Energy	Air Pollution	Hazardous to human health	Communities	Cost to health services of treatment	£
Material Use	Release of hazardous materials	Hazardous to human health	Communities	Cost to health services	£

Table 2 - Economic indicators for transport system sustainability.

2 .4. Human/Social Indicators for Transport System Sustainability

The Human/Social indicators reflect a crucial part of the sustainability analysis. Ultimately, if the impacts on this store of value are overwhelmingly negative, the reasons for the system's existence come into question. The EIRs affected by impacts to the human/social store of value are *employees*, *users* and *community*. The various system processes were again considered, and the main inputs and outputs were assessed for their impacts on the human/social store of value. Assessing the impacts from resource use and waste is relatively straightforward – however evaluating the impact of mobility on sustainability is more difficult.

2 .4.1. The Sustainability Impact of Mobility

The main output of the car fleet is mobility. Its impact on sustainability is encompassed by two separate questions: what is the impact of mobility in general on sustainability, and what is the impact of the mobility provided by the car fleet specifically?

Answering the first question is both beyond the boundary of this study, and impossible to do objectively. It would require examining all the different uses to which mobility is put and assessing the relative benefits of each journey – but this runs into the issue that what is a crucial trip for one person is an unnecessary luxury for another. Furthermore, any given trip can be made using a variety of different modes, so examining the impacts of the trips made does not impart any information on the merits of the transport system under scrutiny.

Thus, the impact of mobility on sustainability was reduced to answering the question: "what is the impact of the mobility provided by the car fleet specifically?" This is achieved by measuring a combination of the quality and quantity (in passenger-km) of mobility provided by the system to assess how well the system is performing, or, in other words, its 'fitness for purpose'. This is consistent with our definition of a sustainable transport system, which requires a system to provide levels of mobility and safety of movement that are equal to what the society considers to be an ideal level.

Quantity of transport is straightforward to measure. As was discussed in 2 .1.2 ('Working Definition'), mobility is considered to be a positive. For quality, different perspectives are considered: that of the *users*, and that of the *community.* For the user perspective, a set of 'service quality' indicators was generated based on the literature to capture the key characteristics of transport system performance. These are based around three basic questions – how long the trip will take (including how soon it can be started), under what conditions will it be taken, and how safe it will be. The indicators are presented in Table 3 below.

A key consideration in the assessment of these indicators is how much control the user has over the indicator. The reason that user control is important ultimately relates back to our definition of a sustainable transport system, which requires "*levels of mobility, accessibility, and safety of movement that are equal to what the society considers to be ideal levels".* The more control a user has over a particular aspect of a journey, the closer that journey will be to '*ideal*' in the user's eyes. For example, if one considers the indicator 'speed', car users (or at least drivers) have some control, but are nominally within the bounds of legal limits, vehicle performance, traffic control measures, and other road users (congestion). They are also restricted in terms of the route they can take by the road infrastructure. Ultimately this means that the *speed* indicator is important and needs to be measured. On the other hand, in the case of the sub-indicators of *cleanliness*, *crowding* and *luggage capacity*, the users have almost complete control: by buying a larger vehicle, taking fewer passengers or cleaning their vehicle more often the users can decide what levels are acceptable to them. As a result, these indicators can be deemed to be at ideal levels. This is not the case for other modes.

An interesting corollary to this argument is that the illusion of control (or of greater control) can be more important in terms of the user satisfaction than actual levels of service (Pellino and Ward, 1998). This is apparent in terms of speed (where the user is far more restricted than might appear) but also in terms of safety, where although the driver has full control over the vehicle (including what safety features are on it), accidents can still happen through the actions of other road users.

There are several issues with assessing the service quality performance of a specific transport system. The first is that the world our society inhabits has been greatly influenced by the types of transport system that have existed within it. As one mode has developed, so has the infrastructure around that mode, and with it societal expectations of what is accessible, and in what time. An example of this would be out-of-town retail parks, which exist largely with the expectation that most people have access to cars. This affects the assessment of a mode's suitability as the more dominant a mode becomes, the more the environment shapes itself around it, making that mode increasingly attractive as time goes by. An analysis of service quality is however still useful, as a means of identifying what characteristics of a system make it attractive, and what aspects are below ideal levels. Although the service provided by a dominant mode may be inflated as a result of its dominance, unless it achieves ideal levels of all service indicators, the assessment is still useful. It is however important to consider this issue when assessing "minority" modes, as these will be negatively impacted by their small market share. A second issue is that the service quality provided by a mode can vary based on the characteristics of that journey. For instance, is it long or short, will there be a need to carry luggage, what are the weather conditions etc. All these factors can influence the suitability of a particular mode over another from the user perspective.

Another issue is that of weighting different indicators. Different people have different priorities, and as a result creating any sort of ranking between the various indicators is impossible to do objectively. The issue is further compounded by the existence of thresholds based on societal norms and expectations. For instance, although safety is obviously more important from a societal perspective than speed or comfort, once a mode is deemed safe enough to travel on, other considerations come to the fore.

An issue specific to private vehicles is differentiating between drivers and passengers. Obviously, many of the benefits of private transportation are reduced for passengers, as passengers typically have far less control over the journey and choice of vehicle.

Finally, the specific purpose of the study will influence the pertinence of different indicators – for instance a study comparing different modes will focus on different aspects than one studying how a single mode is changing over time. Alternatively, a study might focus on the service provided within a given context, such as commuting, shopping, or leisure use. The list of indicators of service quality Table 3 provides can be adapted to suit the questions being asked by the assessment. The metrics also need to be selected accordingly.

Theme	**Indicator**	**Sub-indicator**
How long will the trip take?	Immediacy	Proximity of mode to origin/destination
		Frequency of departure
	Speed	Average speed on route
		Efficiency of route
	Reliability	Variability/consistency of trip time
		Reliability of mode operation
How comfortable is the trip?	Comfort	Access to services
		Quality of services
		Noise
		Temperature
		Crowding
		Cleanliness
		Level of physical/mental effort
		Exposure to outdoors
		Luggage capacity
		Seating availability/quality
		Privacy
How safe will is the trip?	Safety (user)	Killed or seriously injured per km
		Fatalities per km
		All severities
	Security (User)	Exposure/ protection afforded against crime

Table 3 – List of possible service quality indicators

Simply considering service quality from the perspective of the individual user is insufficient. An important aspect of a transport system relates to how well it caters to the needs of the entire community. This means examining the levels of access to the mode based on age, socioeconomic status, and disability, to ascertain whether specific demographic groups are being excluded from its use. The sub-indicators for *equality of modal opportunity* are *affordability*, *modal inclusiveness dependent on age, infirmity and disability*.

2 .4.2. *Other Human/ Social Impacts*

The performance driven assessment of the mobility is not sufficient to fully describe the impact on sustainability. The impacts of the system on the wider community also need to be considered.

Collisions involving cars impact communities through the casualties they cause among both the car users, and other road users implicated in the collisions. Of particular concern are vulnerable road users, such as pedestrians and cyclists. Although not all these casualties are the ‘fault’ of the car driver, they are a direct consequence of cars being on the road and must therefore be considered.

The impacts of air pollutants on human health have been the subject of a number of epidemiological studies. In their review of recent research into the health effects of air pollution, Ren and Tong (2008) conclude that there are “*there are consistent short-term effects of air pollution on health outcomes (hospital admissions or deaths)*” and that there is also evidence linking air pollution to long-term effects. However not all pollutants have been linked to long-term effects. The indicators chosen to describe this impact are “deaths brought forward due to car-related air pollution” and “hospital admissions caused by car-related air pollution”. It could be argued that these impacts could be considered as environmental impacts. However, given that they directly relate to human well-being they are considered as affecting the human/social store of value.

In a similar vein, noise emissions also impact human health. From WHO reports (WHO, 2000 and 2005), a number of health impacts can be identified:

- Disturbed sleep
- Annoyance
- Cardiovascular diseases

- Hearing impairment and tinnitus
- Cognitive impairment
- Impaired communication
- Increased aggression

The difficulties in estimating the health impacts come from assessing the levels of exposure of the population to noise. Furthermore it is difficult to then discern the share of noise disturbance attributable to cars, especially given that a lot of factors affect the impact of car noise, such as the volume of traffic, the times of day when the noise occurs, where it occurs, insulation of nearby buildings, the speeds at which the cars are travelling etc. The volume of traffic is particularly important.

Employment generated by the transport industry also impacts the human/social store of value, by providing opportunities to the communities to support themselves financially. In some instances these also provide other benefits, such as training, but for the purposes of this study these are not considered. Table 4 provides the complete list of human/social indicators for assessing the sustainability of a transport system. The impacts of each process have been aggregated according to the top-level inputs and outputs identified in Figure 6.

Input/ Output	**Impact**	**Issue**	**EIR**	**Indicator**	**Metric**
Workforce	Employment	People can support themselves	Employees	People in employment	nº of ppl
Energy	Air Pollution	Hazardous to health	Communities	Deaths brought forward due to car-related air pollution	nº of ppl
				Hospital admissions caused by car-related air pollution	nº of ppl
Mobility	Mobility is provided	People are able to travel	Users	Passenger-km produced by car fleet	Passenger -km
	Service quality	People are able to perform trips under certain conditions	Users	Variety of service quality indicators	-
		The mobility needs of the community are met	Communities	Equality of modal opportunity indicators	-
	Noise	Hazardous to health	Communities	Morbidity and mortality impact of car noise	nº of ppl
	Accidents/ Injuries	Resulting injury	Communities	Number of killed or seriously injured	nº of ppl.

Table 4 - Human/ Social indicators of transport system sustainability.

2 .5. Discussion

Applying the Darton-Tahir framework to a transport system showed some limitations of the method, arising from some of the new challenges presented by the transport system in contrast to the case study performed by Tahir on palm oil production. This can principally be attributed to two major differences between the systems: one is product based and under centralised management, and the other is service based and under decentralised management. However, the strengths of the method also became apparent, and the indicator set that was generated is more robust as a result of its use.

The choice of definition of sustainability for the study showed that simply falling back onto the Brundtland definition was insufficient for the purposes of fully understanding what sustainability means for a transport system. It was deemed necessary to further research this question by undertaking an analysis of the goals and constraints of the system with respect to sustainability. Various definitions were tailored to better suit the system characteristics. This is partly due to the fact that a transport system is service based – a first step was to identify that the impacts of the service itself are relevant to sustainable development, and not only the costs involved in providing it. Therefore it is important to build into the definition a statement of the goals of the system under scrutiny, i.e. the service it is designed to provide. Gudmundsson and Hojer (1996) concur: "*the transport system and the services it provides cannot be separated*".

This finding leads to a second important point, analysing the impacts of the service provided on the sustainability domains, an aspect which is often overlooked. Difficulties arise when different systems are competing to provide the same (or similar) service(s). In this situation, the impacts of the system become difficult to discern – what needs to be measured is no longer simply the impacts of the service provided, but the impacts of that service being provided by the system under scrutiny rather than by another. For the purposes of the human/social impacts of a transport system, this was achieved via a rating of service quality comprising of indicators measuring the performance of the system relative to the user and the community. These are comparable to a measure of ‘fitness for purpose’ of the system. Some sources from the literature also consider the performance of transport systems in their sustainability analyses, although not necessarily in the same ways. Whitmarsh et al (2009) generate criteria for sustainable transport based on stakeholder engagement. Interestingly, the social sustainability criteria suggested by the “citizen” (as opposed to expert) panel include “reliable/regular, fast, fun, enjoyable, choice, clean, aesthetic [amenity/utility]” which reinforces the importance of these quality aspects in a sustainability analysis. Kennedy (2002) includes ‘level of service’ in his sustainability assessment of public and private transportation in the Greater Toronto Area, mentioning aspects such as comfort and personal security, but focusing on speed and access (in terms of ability to access different parts of the city) as quantifiable measures. The SUMMA project (Walker et al., 2006) also includes an assessment of service characteristics, but does not include them in their main list of indicators.

A third point is that a number of the impact receivers identified by the method, namely the *users* and, to a lesser extent the *vehicle manufacturers* (insofar as they are capital providers), can also act as impact generators via their decisions. This is due to the car fleet being largely decentralised – a number of stakeholders exert an influence on the system, whilst simultaneously being affected by the impacts of other stakeholder decisions. For example, *user decisions* regarding transport will affect not only other stakeholders but the *users* themselves. *Users* are therefore receivers of certain impacts, such as localised pollution or congestion, but also influence their creation. The framework does not explicitly investigate feedback loops such as this. However, it did not encounter any problems as a result of this loop being present. The *users* form a unique stakeholder group, in that the impacts they generate via *user decisions* are the emergent result of a multitude of individual decisions. This is a very important consideration and fundamental to understanding why sustainability in transport has proved hard to bring about, due to how such an entity reacts to feedback. An individual user is unlikely to attribute congestion to their particular presence on the road, and yet it is the result of many users opting to be on the road that creates the congestion. It is worth noting that neither *users* nor *user decisions* form homogenous entities. It is possible that individuals within *users* suffer more from certain impacts than others, but do not have a sufficient influence on overall *user decisions* to improve their situation. This is very different from the situation in palm oil industry, which is under centralised management.

User choice can be described in terms of a utility function, which values the outcomes of particular decisions for the user – in this case, transport related decisions – according to the relevant criteria. Each utility function is unique to the user and is a reflection of his preferences. However, the criteria on which that decision is based are common to all, even if they are valued differently – they are the characteristics of the system. To fully understand the system, the impact of user choice must be considered. An important aspect of the service quality indicators is that they represent some of the criteria on which users might base their decisions. Given the extent to which user choices affect the system, and hence its sustainability, this is an important consideration.

The indicator set produced by the method addresses a lot of the issues with other sustainability assessments. The set is comprehensive without being overwhelming. This is achieved through an assessment of each process occurring within the system in terms of both the resources consumed and the wastes and products. The relevance of the indicator set is ensured by linking each impact to an issue or issues affecting specific stakeholders within the system. This addresses the problem raised by Beherends et al. (2008):

> *“Existing indicator sets either fail to reduce the complexities and grapple with the interdependencies in urban transport or are not applicable to the actors involved (impact indicators of an end state vision).”*

Moreover, the indicator set was generated by applying a clear methodology, linked to a carefully considered definition. Many sustainability analyses generate indicators on a seemingly ad hoc basis, sometimes focusing on just one aspect such as CO2 (Grimes-Casey et al., 2009) or on several but with no clear method by which they were selected (Kennedy, 2002, Fedra, 2011).

In terms of the criteria for sustainable mobility indicators proposed by Gudmundsson (2003), our indicator set addresses mainly the first (providing a comprehensive picture of the current impacts from mobility), with some attempt at the second (identifying the causal factors) through a consideration of the Internal Impact Generators. It will be shown in chapters 3 and onwards that a practical application of the indicators to a real case study leads to causal factors being identified, but this is not a process that is explicitly built into the methodology.

The approach taken with this study has been to consider mobility to be a positive, and to assume that the greater the provision of mobility the better. However, some researchers opt to focus more on access (to jobs, goods and services) than on mobility, such as Lucas et al. (2007), and the SUMMA study (Walker et al, 2006). Gudmundsson and Hojer (1996) note that they “*are not convinced that mobility is an undeniable good, and not a condition compulsively forced upon the individual by social and physical structures*”, suggesting that “*if access can be provided in some other fashion than by the physical movement of people and goods, this may be the most promising option in the longer term.*” Banister (2008, 2011) argues that t efforts towards sustainable mobility should focus on the reduction of demand, both in terms of distance and number of trips. He links the issue to urban form, where the structure of the built environment affects the demand for transport. Whilst we would agree with the above points, it is to some extent a question of boundaries – this study focuses purely on the provision of transport, whereas the 'accessibility' approach includes the larger question of the uses to which it is put. Reducing demand for a particular good or service will reduce the resources consumed in its provision, and hence improve its sustainability. It is ultimately society's decision how to manage its resources and consumption.

Some indicators are conspicuously absent from this indicator set, namely those that relate to the supporting infrastructure of the car fleet. This was left outside the boundary for reasons explained in 2 .1.3, but was included in studies such as SUMMA (Walker et al, 2006). The impacts include the costs of maintaining the infrastructure, as well as its impact in terms of the livability of public spaces, aesthetic impact, and land use.

2 .6. The UK Car Fleet Between 1995 and 2005 - a Case Study

Having generated a set of indicators to assess the sustainability of a transport system, it was deemed both necessary and interesting to test the indicator set with a case study. The study considers how the sustainability of the UK private and light goods[1] fleet compares across two years, a decade apart. The years under scrutiny, 1995 and 2005, were chosen for several reasons. They relate to a comparatively modern period, during which awareness of sustainability issues has increased significantly. This is reflected in increasing regulation concerning environmental impacts and a greater focus on transport related issues such as congestion and safety. Data availability was also a consideration: the further back one goes, the less data are available, and very recent data tend to take a year or two to be made publicly available.

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[1] In the context of this study, the private and light goods vehicles will occasionally be collectively referred to as the 'cars'. Where the distinction is needs to be made, body type cars are referred to as 'private cars'.

Despite the study period being chosen to minimise this problem, the indicator set proved challenging to populate as a result of patchy data availability. As a result, some indicators were not measured, and in some cases proxy indicators were found that only provide limited insight into the issues being measured. It is hoped that by creating a need for certain types of data, in the future more will become available. Dalal-Clayton and Bass (2002) state that one of the first steps on the path to sustainability assessment is the creation of a framework for data collection, and note that often areas with the greatest need for sustainability assessment are also the most lacking in available data with which to perform such an assessment. Whilst transport as a field is increasingly aware of sustainability issues, there are still gaps in data availability that need to be filled.

The results of the study, as well as a description of how the indicators were calculated and a discussion of how the sustainability of the car fleet is changing are presented in Chapters 3, 4, 5 and 6.

Chapter 3

Environmental Impacts of the UK car fleet

3 .1. Introduction

The manufacture, operation and disposal of the vehicles in the car fleet generate a variety of impacts, broadly linked to material and energy use. Due to the complex supply chain of a vehicle lifecycle, there are some boundary issues. For a start, not all new vehicles added to the fleet are manufactured in the UK – many are imported. Secondly, those vehicles manufactured in the UK may have imported parts or materials from elsewhere. The approach has been to account for all the vehicles added to the UK car fleet, regardless of where they are produced, on the basis that environmental impacts occur irrespective of location, and that the levels of mobility produced depend on the influx of new vehicles. It should be noted however that in 1995 UK vehicle production accounted for 87.3% of new vehicles, whereas in 2005 only 66.5% of new registrations would be covered by UK production (production data from ONS, 2006).

3 .2. Changes to the Car Fleet

There were significant changes to the car fleet over the time period considered, both on a fleet level and an individual vehicle level. These changes affected the environmental impact of the system.

3 .2.1. *Fleet trends*

On a fleet level, several trends emerged. Firstly, there was a sharp increase in the number of vehicles registered (Table 5), both annually and in total. This lead to a 16.2% rise in annual vehicle-km (395.6 to 459.8 billion). The composition of the fleet also changed with respect to engine type: in 1995, only 27.73% of new vehicles, and approximately 16.9%[2] of total vehicles were diesel, compared to 45.6% of new vehicles and 27.9% of total vehicles in 2005. This increase in diesel vehicles is reflected in the respective quantities of petrol and diesel fuel consumed, as seen in Table 6.

	1995	**2005**	**Change**
Private cars registered at end of year	20505	26,208	27.8%
Light goods registered at end of year	2217	3,019	36.1%
Total private and light goods end of year	22,722	29,227	28.6%
New private and light goods registrations	2,024	2,604	28.7%

Table 5 - Vehicle registrations, thousands (DfT, 2007a).

	1995	**2005**	**Change**
Petrol	1,020	863	-15.4%
Diesel	210	422	100.6%
Total	1,230	1,285	4.4%

Table 6 - Fuel energy consumption, in GJ x 10^6.

It should be noted that the overall increase in fuel energy (4.45%) consumption does not match the increase in vehicle-km (16.2%). This shows that the energy efficiency of UK private and light goods vehicles improved over the study period. This may be partly explained by the 1998 commitment signed by auto manufacturers to reduce CO_2 emissions by 2008, (Table 7). Table 7 also details some of the other relevant legislation adopted over the study period, which will be discussed later.

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[2] The exact figure not being available from the DfT Transport Statistics for 1995 due to differences in classification methodology, this was estimated by applying the fraction of cars by body type with diesel engines in 1995 to the number of cars by tax class in 1995, and applying the fraction of light goods vehicles by body type with diesel engines in 2005, to the number of light goods vehicles by tax class in 1995.

Regulatory Timeline	
1993	UK fuel duty escalator – inflation + 3%
1996	“Euro 2” emissions standards (regulates CO, NO_x, unburnt hydrocarbons, and PM) come into force (EC, 2007)
1997	UK fuel duty escalator – inflation + 6%
1998	ACEA signs voluntary agreement to reduce fleet CO_2 emissions to 140g/km by 2008
Sep. 2000	EU End of Life Vehicle directive adopted by EU
Nov. 2000	UK fuel duty escalator – ended
Dec. 2000	“Euro 3” emissions standards come into force
Nov. 2003	UK regulations implementing first part of the ELV directive come into force. Restrict the use of heavy metals in new vehicles and components, introduce a Certificate of Destruction (CoD) for ELVs and ‘take-back’ arrangements for vehicles put on the market from 1 July 2002. (BIS, 2010)
2005	“Euro 4” emissions standards come into force
Mar. 2005	UK regulations implementing further parts of ELV directive come into force. Set reuse, recycling and recovery targets for ELVs from 2006 onwards.

Table 7 - Relevant UK and European Commission Legislation.

3 .2.2. *Vehicle Trends*

The vehicles themselves also changed over the time period, not only in terms of engine efficiency and type. Broad trends can be observed in vehicle weight and composition, which both have consequences in terms of impacts on the environmental store of value.

Several sources were considered with respect to average weight. In an update to the EC regarding progress towards targets for CO_2 emissions (European Commission, 2006), the European Automobile Manufacturers Association (ACEA) stated that the average weight of a new vehicle went up approximately 14.5% between 1995 and 2004. The average weight of a new vehicle in the UK in 2004 was given as 1384kg (no figure was available for 2005). Information on weight was also found in a report detailing a life-cycle analysis of different vehicle disposal methods commissioned by the Environment Agency and the

DTI (Ecobalance UK, 2002). This surveyed major vehicle manufacturers for data on weight and composition, and then generated an average using the data on new registrations in the UK in 1996. This produced an average weight of 1043kg. If one applies the 14.5% increase in weight to this estimate, the result is 1194kg, almost 200kg less than the ACEA estimate for 2004. A decision was nevertheless made to use these values for the study since they were based on actual new registration data for the UK. However, both sources are legitimate, and as a result it should be noted that impacts due to vehicle weight could be approximately 16% higher. The Japanese Automobile Manufacturers Association (European Commission, 2006) suggests that improved safety features account for approximately 27% of the weight increase, with the remaining increase being due to changes in market demand (for instance, the average engine size has increased). The increase in weight has several consequences in terms of sustainability. First, it causes significant growth in the embodied energy in the fleet. This impacts emissions and resource depletion. Secondly, it impacts material use (resource depletion) and potentially causes waste issues. Thirdly, weight increase offsets gains in fuel efficiency (impacting emissions and resource depletion). Not all impacts are negative; additional safety features have contributed to the decrease in the number of users killed or seriously injured (see chapter 5). Similarly, there has been an increase in the comfort and performance of vehicles, which impacts the quality of the mobility provided.

Ecobalance UK (2002) also suggested a material composition for vehicles in 1996 based on a survey of the dominant manufacturers in the UK, which was used as the basis for a typical material composition of a new vehicle in 1995. The material composition for a typical new vehicle in 2005 was based on the IMPRO-Car study (Leduc *et al.*, 2008), which provided estimates for generic vehicles in 2004. The main changes between the two vehicles were larger proportions of iron and steel in the 1996 vehicle, and of plastics in

2004. This is in line with trends in composition discussed by Giannouli *et al.* (2007). It is interesting that despite a trend towards lighter materials, average weight still increased. The other large difference between the two compositions was that the amount of coolant in the 2004 vehicle was almost twice that in the 1996 vehicle. This could be explained by the increasing use of air conditioning systems. The final compositions used in the study are shown in Table 8. These were modified to be consistent with the GREET model (see 3 .3) used to calculate emissions and energy use. The category titled ‘average plastic’ was used due to the specific plastics from the material compositions not being available in the model. Changes in composition affect the energy required to produce the vehicles, the weight, and also the ease with which they are recycled.

	1995	**2005**
Steel	58.1%	55.0%
Stainless Steel	0.0%	0.0%
Cast Iron	13.8%	10.4%
Wrought Aluminum	1.8%	1.8%
Cast Aluminum	3.7%	3.7%
Copper/Brass	1.2%	0.7%
Magnesium	0.04%	0.04%
Glass	3.4%	3.1%
Average Plastic	10.0%	18.0%
Rubber	2.4%	2.0%
Platinum	0.0001%	0.0001%
Others	5.6%	5.3%

Table 8 - Typical material compositions of new vehicles (excluding fluids and battery) used in GREET.

3 .3. Life Cycle Energy and Emissions

Both embodied emissions and direct emissions are considered in the study. Every year, cars are added to the fleet, creating an emissions and energy burden through their manufacture, maintenance and eventual disposal. Simultaneously, the fleet produces emissions as a result of the vehicles being driven. The fuel consumed also carries embodied emissions. Despite the fact that the majority of vehicles added in a specific year will not be scrapped until 10-15 years later, it was decided to account for their full life cycle, since they represent a future burden.

The GREET[3] model was used to calculate embodied energy and emissions in the life cycle of vehicles added to the fleet. Its principal inputs are material composition and weight. The model is based around two cycles: the fuel cycle and the vehicle cycle. The vehicle cycle accounts for emissions from material and energy use during the vehicle life cycle, resulting from manufacture and some aspects of operation and disposal. The processes included in operation are the replacement of parts and fluids, such as windscreen fluid, oil, batteries and tyres, whilst the processes included in disposal are dismantling and shredding. Energy use from recycling is attributed to the industry making use of the recycled product. Similarly, energy from energy recovery is not included in the model. The direct fuel use (i.e. the Well-to-Wheels, or WTW component) resulting from operation of the vehicle is modeled in the fuel cycle, which also includes the primary energy and emissions embodied in different energy carriers (fuel, electricity, etc.). For this study, only the vehicle cycle was used – data on the WTW component were found elsewhere due to GREET being a US model. However, the vehicle cycle model refers back to the fuel cycle model when calculating the emissions caused by energy use during the vehicle life cycle. Some parameters such as the electricity generation mix were therefore changed in the fuel cycle

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[3] Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) Model, Argonne National Laboratory, USA.

model to better reflect the UK situation (this was adjusted according to DUKES, DECC, 2006). Nevertheless, many of the default assumptions on energy requirements for processes such as material extraction and processing and energy production pathways were used. The material compositions used are those displayed in Table 8, and the average weights are those discussed in section 3 .2.2. Other important inputs are the proportions of recycled materials in the vehicles - these were taken from BIR (2010) and UNEP (2011), and are displayed in Table 9.

	Virgin Material	**Recycled Material**
Steel	46.0%	54.0%
Wrought Aluminum	64.0%	36.0%
Cast Aluminum	64.0%	36.0%
Lead	49.0%	51.0%
Nickel	59.0%	41.0%

Table 9 - Proportions of virgin and recycled material in vehicles.

In terms of the WTW component, direct emissions data were taken from the Department for Transport for the private and light goods class (DfT, 2007a). Data for impact of embodied emissions in fuel, or Well-To-Tank, were taken (on a per unit of fuel basis) from the CONCAWE report (Edwards *et al.*, 2007), and multiplied by the fuel use data in DfT (2007a).

The results are displayed below. Table 10 shows the impacts associated with direct emissions due to fleet operation. Table 11 shows the impacts associated with the embodied emissions and energy in the fuel used (Well-to-tank). Table 12 shows the impacts associated with the embodied energy and emissions in the vehicles added to the fleet. The impact categories discussed in Chapter 2 were not all calculated. For instance, Leduc *et al.* (2008) note that whilst eco-toxicity is an important impact, it is difficult to quantify the amount of toxic releases, due to the large number of substances, lack of agreement on toxicity potentials, and lack of data on emissions factors from processes causing releases.

Therefore this category was omitted. The most salient point regarding toxicity is the large decrease in direct emissions of lead, which dropped 99.8% to 1.6 tonnes in 2005. Similarly, abiotic depletion was not calculated for the vehicle life cycle. Firstly there was a lack of sufficiently detailed data on the more exotic materials used, and secondly the largest material categories (the metallic fraction) are recycled. Similarly insufficient data were found on the impacts of material extraction. Finally, the emissions data from GREET and DfT (2007a) did not include any pollutants linked with stratospheric ozone depletion.

To calculate the indicators, characterization factors were mostly taken from Guinée *et al.* (2002) and applied to the data on emissions, except for the Photochemical Oxidation Potential (POCP) factors of NO_x and 'average' Volatile Organic Compounds (VOCs), which were taken from Leduc *et al.* (2008). Typically factors are not given for NO_x since the exact ratio of NO and NO2 greatly impacts the ozone formation potential (at high NO_x concentrations, NO oxidises to NO_2, inhibited O_3). However, Leduc *et al.* (2008) reviewed several studies that suggested the overall impact of NO_x on ozone formation was nevertheless positive. It is likely that the POCP indicators is more accurate for 2005, since overall NO_x levels dropped significantly over the time period. This is discussed further in Chapter 5. Not all the sources provided the same information on emissions – the DfT data on direct emissions from vehicles included lead and benzene, which were not included in the GREET model. The GREET model also describes emissions of in the form of a single category of unspecified VOC's. Although a fraction of these were identified as being ethanol and methanol releases from windshield wiper fluid, the rest were indeterminate, meaning that an accurate characterization could not be applied.

The results show that whilst impacts due to direct emissions broadly fell, the impacts from the rest of the vehicle life cycle rose. This is due to a combination of progressively stricter tailpipe emissions standards coming into force and reducing direct emissions, and increases in both the number of new registrations and vehicle weight causing greater embodied emissions. There was also small increase in fuel consumption over the period, which led to a slight increase in embodied emissions from fuel (it should be noted that total vehicle mileage increased 15.8% over the study period). The largest reductions were in emissions of pollutants covered by the EURO standards. Emissions of CO_2, which were only regulated by a voluntary agreement, rose 6% over the period, although there was an 8.5% drop in CO_2 emissions per vehicle-km. This improvement was offset by the rise in overall mileage.

	1995	2005	Change	
Global Warming	81800	86700	6.0%	Thousand tonne CO_2-eq.
Ozone Layer Depletion	-	-	-	Thousand tonne CFC-11-eq.
Photochemical Oxidation	554.1	199.1	-64.1%	Thousand tonne ethylene-eq.
Acidification	516.9	193.5	-62.6%	Thousand tonne SO_2-eq.
Eutrophication	90.7	35.5	-60.9%	Thousand tonne PO_4-eq.
Particles PM10	29.8	24.0	-19.5%	Thousand tonne
Primary Energy Use	1229.9	1284.7	4.4%	GJ x 10^6

Table 10 – Impact of direct emissions caused by vehicles in private and light goods fleet.

	1995	2005	Change	
Global Warming	16199.3	17122.5	5.7%	Thousand tonne CO_2-eq.
Ozone Layer Depletion	13.5	14.1	4.4%	Thousand tonne CFC-11-eq.
Photochemical Oxidation	61.0	62.1	1.8%	Thousand tonne ethylene-eq.
Acidification	223.2	223	-0.1%	Thousand tonne SO_2-eq.
Eutrophication	18.2	18.8	3.3%	Thousand tonne PO_4-eq.
Particles PM2.5	5.7	5.8	1.6%	Thousand tonne
Primary Energy Use	176.4	188.3	6.7%	GJ x 10^6

Table 11 - Impact of embodied emissions in fuel used by vehicles in private and light goods fleet.

	1995	2005	Change	
Global Warming	12007.9	17972.3	49.7%	Thousand tonne CO_2-eq.
Ozone Layer Depletion	-	-	-	Thousand tonne CFC-11-eq.
Photochemical Oxidation	21.1	28.8	36.5%	Thousand tonne ethylene-eq.
Acidification	41.7	58.2	39.4%	Thousand tonne SO_2-eq.
Eutrophication	2.4	3.5	44.9%	Thousand tonne PO_4-eq.
Particles PM2.5	8.6	13.0	50.8%	Thousand tonne
Primary Energy Use	162.8	241.8	48.5%	GJ x 10^6

Table 12 – Impact of embodied emissions in vehicles added to the private and light goods fleet.

3 .4. Disposal and Waste

3 .4.1. Waste from Manufacture

In addition to the energy use and emissions resulting from manufacture, there are additional impacts in terms of other wastes generated. From the SMMT sustainability reports (SMMT, 2007 and earlier annual reports), the average waste to landfill per vehicle produced, and average water use per vehicle produced were identified for the year 2005 and the year 2000. Earlier data were not available since the reports have only existed since 2000. Even over the five years, clear progress was achieved. It is likely that the existence of sustainability monitoring helped to drive the changes. These per-vehicle rates of waste and water were multiplied by the number of new registrations in 1995 and 2005 to provide an estimate of waste and water use during vehicle manufacture during the two years (Table 13). It should be noted that these data were produced by the SMMT, and describe the situation among UK producers. Recall that UK production does not cover the total number of new registrations in the UK. It was assumed that the situation would not be significantly different among foreign manufacturers.

	1995 (2000)	**2005**	**Change**	
Waste to Landfill Per Vehicle	40.3	14.5	-64.0%	kg
Total Waste to Landfill	81.57	37.7	-53.7%	Thousand tonnes
Water Use Per Vehicle	5.3	3.2	-39.6%	m^3
Total Water Use	10727.2	8331.5	-22.3%	Thousand m^3

Table 13 - Waste to landfill and water use during manufacture of vehicles entering the fleet, 1995 and 2005, adapted from SMMT (2007)

3 .4.2. Waste from Operation

The operation of the fleet requires not only fuel, but also the replacement of parts and fluids as these wear. Data on waste tyres and disposal were obtained from the Used Tyre Working Group (UTWG, 2009) and are presented in Table 14. Several points should be noted regarding the data. Firstly, the UTWG statistics only detailed estimates for tyres used by private and light goods vehicles as far back as 1999. Secondly, when plotting the data for arisings between 1999-2005, an oscillation was observed. Given that arisings of waste tyres would not be expected to fluctuate year on year, a linear trendline was calculated. Both the data and the trendline are displayed in Figure 9. The trendline allows estimates of tyre waste arisings to be made, which correlated more closely with annual vehicle-km (correlation coefficient of ~0.98) than the raw data (such a correlation would be expected, given that tyres wear with use). They were thus deemed to provide a better reflection of tyre waste. Additionally, use of the trendline allowed an estimate to be calculated for 1995.

	1995	**2005**	**Change**
Used Tyre Arisings	178.5*	221.8*	24.2%
Landfill	61.4	15.2	-75.3%
Recovered	65.6%	93.2%	
of which			
Reuse	18.2%	19.6%	
Retreading	39.0%	12.7%	
Recycling	13.0%	35.8%	
Landfill Engineering	4.6%	13.0%	
Energy Recovery	25.3%	18.9%	
*Based on estimates from trendline			

Table 14 - Used tyre arisings due to replacement, and breakdown of recovery options, 1995 and 2005, thousand tonnes.

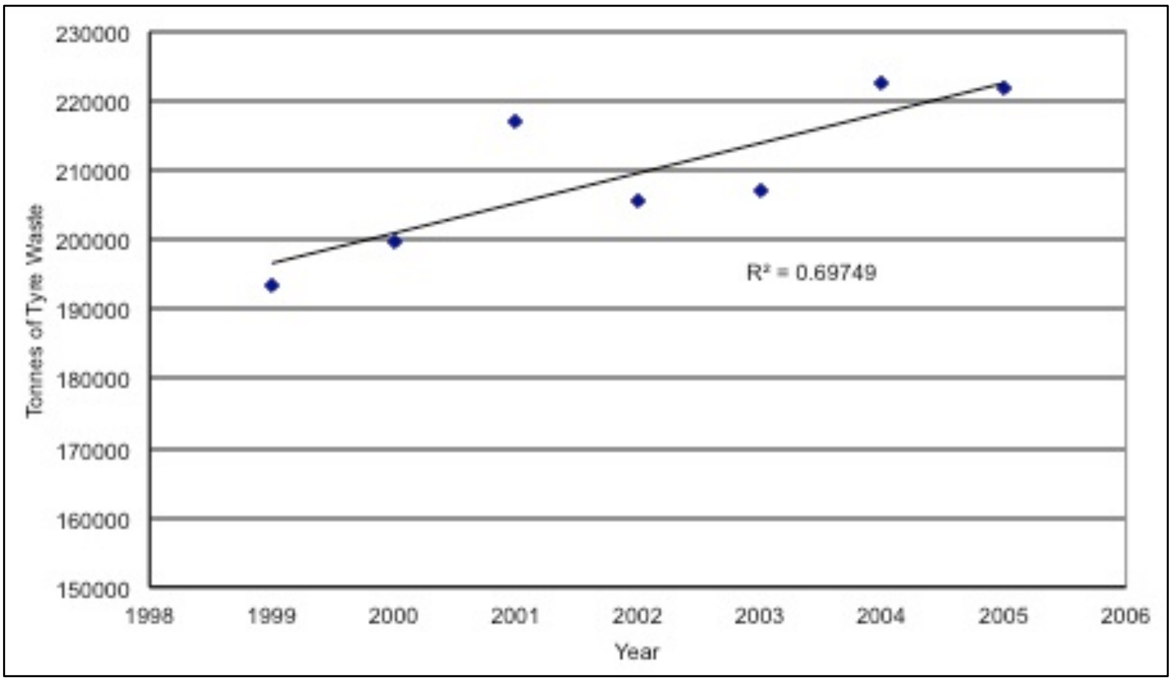

Figure 9 – Arisings of waste tyres resulting from replacement, tonnes. Data from UTWG, 2009.

It can be seen that the quantity of tyres recovered and treated increased significantly over the period, so that the total waste to landfill reduced despite an overall increase in waste tyres. This is due to the EU Landfill Directive, which came into force on the 16th July 1999. This banned the landfilling of whole used tyres from 2003 and of shredded used tyres from 2006. The only allowed disposal of tyres to landfill is in the form of landfill engineering.

TRL (2003) performed a thorough analysis of End-of-Life vehicle (ELV) arisings in 2000, as part of which they examined the consumption of parts and fluids by the private and light goods fleet. Using a combination of estimates of average part life and industry sales data, they provide an estimate of the parts consumed per year, per vehicle. They also provide estimates of average weight, which can be used to estimate the annual weight of waste produced per vehicle. This was multiplied by the number of vehicles in the fleet in 1995 and 2005 to produce the estimates of waste arisings in Table 15. Unfortunately, only limited data on the treatment of these parts and fluids were available. TRL (2003) estimated that approximately 90% of batteries are recycled. Between 87.3% and 97.3%

(two studies were considered) of waste oil was collected, most of which (between 93% and 100%, depending on the study) was cleaned and used as fuel. GHK (2006) states that over 98% of ferrous and non-ferrous metals are recycled, so the majority of the metallic fraction of the above parts does not go to landfill. Given limited data on the composition of many of the parts however, it is difficult to accurately estimate the fraction of waste going to landfill. However, by assuming 90% recycling for batteries, 97.3% recovery for oil, and then applying overall recycling, reuse and recovery rates for ELVs for 1995 and 2005 (76% and 81% - these are discussed in more detail in the following section) a rough approximation can be made. This yields a waste to landfill of 59.4 thousand tonnes for 1995 and 62.5 thousand tonnes for 2005, an increase of 5.2%.

		1995 estimate	**2005 estimate**
Batteries		44.7	57.5
Exhaust system		63.4	81.6
Windscreen		5.9	7.5
Oil Filter		10.1	13.0
Shock absorbers		3.3	4.3
Wiper blades		7.2	9.3
Front brake pads		0.8	1.0
Rear brake pads / shoes		0.6	0.8
Brake discs/drums	Front	27.0	34.8
	Rear	23.4	30.1
Clutch		20.3	26.1
Catalytic converter		1.4	1.8
Air filter		4.4	5.7
Fuel filter		1.0	1.2
Engine oil		109.4	140.7
Engine coolant		43.4	55.8
Brake fluid		4.2	5.3
TOTAL		370.6	476.7

Table 15 - Arisings of replacement parts and fluids, 1995 and 2005, thousand tonnes.

3 .4.3. Waste from Disposal

End-of-Life vehicles (ELV) constitute an important waste stream. The number of ELV can be estimated using vehicle-licensing statistics from the Driver and Vehicle Licensing Agency (DVLA), available in DfT (2007a), by the following calculation:

$$\begin{aligned}ELV_{yearx} = {} & \text{Vehicles licensed at end of year}_{year\ x-1} \\ & + \text{New Registrations}_{yearx} \\ & - \text{Vehicles licensed at end of year}_{yearx}\end{aligned}$$

This is not an entirely accurate estimate. Firstly it does not account for unlicensed vehicles. Secondly, it does not account for any unrecovered stolen vehicles not entering the waste stream (as these would have lost their licenses but not be ELV). Finally, it does not account for any exports. However, TRL (2003) performed a more detailed calculation for the UK ELV arisings in 2000 using insurance industry data on stolen vehicles and ONS data on exports, and found that the number of unrecovered stolen vehicles and exported vehicles only accounted for approximately 80000 units, which amounts to less than 4% of the lapsed licenses. The impact from theft and exports can thus be reasonably discounted. Estimates of ELV arisings are presented in Table 16.

	1995	**2005**	**Change**	
Number of ELV	1970	2030	3.0%	Thousand ELV
Weight per ELV	910	955	4.9%	kg
Fraction Recycled, reused or recovered.	76%	81%	6.6%	-
Weight of ELV (total)	1792.7	1938.6	8.1%	Thousand tonne
Waste to Landfill	430.2	368.3	-14.4%	Thousand tonne

Table 16 - ELV arisings based on lapsed licenses and waste to landfill, 1995 and 2005.

ELV are a mix of so-called natural ELV (cars at the end of their lives) and premature ELV (cars being scrapped due to accidents). Thus they constitute a very different set of vehicles than those entering the fleet, being mostly much older. The average life span for a vehicle is 10-15 years, so an ELV in 2005 would have typically been manufactured in the early 1990's. GHK (2006) estimates the average weights of ELV in different years (available in Annex 2 of their report). Due to an overall upward trend in vehicle weight, these weights are significantly lower than the weights of new vehicles. The weights are presented in Table 16. It is interesting to compare the estimated weight of an ELV in 2005 (955kg) to our estimate of the weight of a new vehicle in 1995 (1043 kg). One would expect these numbers to be quite similar, given that a new registration in 1995 would be nearing the end of its life in 2005. In fact, when considering the impact of premature ELV, which would increase the average weight somewhat, the two numbers should be very close. As it is, there is a discrepancy of around 9%, which is quite large given the magnitude of changes in weight being discussed. This has to be attributed to the fact that both estimates came from different sources, and that there is a lot of uncertainty in this area.

The proportion of a vehicle being recycled, recovered, or reused also changed over the time period. GHK (2006) collects estimates from two UK studies, both focused on the situation in 2000. The ACORD study estimates overall recycling, reuse and recovery of 76% in 1997 and 80% in 2000. The TRL study (TRL, 2003) however produces an estimate of 76.9% for 2000. GHK (2006) notes that the ACORD study could be considered optimistic, and that the situation in 2005 would only just be approaching 80%. This concurs with a report for the European Parliament (EP, 2007), which estimates recycling, reuse and recovery rates in the UK of 81% in 2004. The estimate of 76% for 1997 seems plausible, and is supported by IEEP (1996), which estimates recycling and recovery in the UK at 77%. The two estimates used were therefore 76% for 1995 and 81% for 2005. The results are presented in Table 16. It can be seen that the increase in recycling and recovery has mitigated the increase in vehicle weight and scrapped stock.

Much of this improvement can be explained by the introduction of the European Commission End of Life Vehicle directive in 2000, which was introduced in UK law in two parts. The first, introduced in 2003, imposes the free 'take-back' by manufacturers of vehicles put on the market from 1 July 2002 onwards when they arise as ELVs (should they have no value when scrapped). It also prohibits the use of heavy metals such as lead, mercury, cadmium and hexavalent chromium, except for specific uses (for instance batteries). Finally it introduces a requirement for a Certificate of Destruction (CoD) for ELVs, and the licensing of authorised treatment facilities for disposal. The second part was introduced in 2005, and sets recycling and reuse targets of 80% and recycling and recovery targets of 85% for 2006 (recovery refers to the process of burning waste for primary energy). Further, more stringent targets are set for 2015. It also mandates free take back of all ELVs from 2007 (BIS, 2010).

A final point should be made regarding methodology. The general approach for the study has been to consider inputs and outputs to the fleet on an annual basis: vehicles added per year, fuel and part use per year, and ELV per year. However, when calculating energy use and emissions, a life cycle approach was applied to the vehicles added to the fleet. This means that energy use and emissions from parts and disposal was calculated for the new vehicles added in a year, rather than for the actual parts consumed, and ELV in that year. This inconsistency is due to difficulties with the use of the GREET model.

Chapter 4

Human/Social Impacts of the UK car fleet

This chapter will present the impacts of the car fleet on the human/social store of value, in 1995 and 2005, and discuss how the indicators were calculated. First the contribution in terms of quantity of mobility is considered, followed by an assessment of the quality of mobility, from both a *user* and a *community* perspective. Finally, the impact of automotive industry employment is considered.

4 .1. Mobility Provided by the Car Fleet

The primary purpose of the car fleet is to provide mobility. As such, one of the most important human/social impacts of the fleet is the quantity of mobility it provides. This has increased over the study period (Table 17), especially the mobility provided by light goods vehicles, which has seen the most growth. The data were taken from DfT (2007a). Interestingly, the amount of km per vehicle on the roads has actually decreased, although the National Travel Survey (DfT, 2006d and ESDS 2010a,b) shows that the average distance driven per year has increased (by approximately 1.6%). This suggests that the growth in vehicle-km[4] (v-km) is due to an increase in users rather than an increase in the distance each user is driving. Users are benefiting less from vehicle ownership in terms of quantity of mobility. However, the car fleet is providing mobility to a greater number of people. This is a clear example of how different indicators can run against each other – from an environmental perspective increased fleet size and mileage has meant more

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[4] Vehicle-km denotes the distance covered by vehicles. Similarly, passenger-km denotes the distance covered by passengers. It equates to the number of vehicle-km multiplied by the average number of occupants.

emissions, material use and energy use, whilst from a human/social perspective, more users have benefited from car-based mobility.

	1995	2005	Change
Private Car Vehicle-Km	3511	3972	+13.1%
Light Goods Vehicle-Km	445	626	+40.7%
Total Vehicle-Km	3970	4598	+15.8%
Total Passenger-Km	6272.6	7264.84	+15.8%

Table 17 - Mobility provided in terms of Passenger and Vehicle-km (100 million)

4 .2. Service Quality and Equality of Modal Opportunity

In Chapter 2, the service quality was considered in terms of three aspects – overall speed, comfort, and safety - which are completed by a fourth, equality of opportunity. A number of the indicators discussed in Chapter 2 (such as *privacy, security,* and *access to services*) are better suited to cross-modal comparisons than to trends within a single mode over time. Other indicators, such as *efficiency of route,* or *proximity of mode to point of origin and destination* are difficult to evaluate quantitatively on a national scale, so proxy indicators were found.

4 .2.1. How long will the trip take?

The time the journey takes is a function of a number of elements linked to the connectivity of the transport network used: proximity of the nearest node in the network to the journey origin and to the destination, frequency of departure, speed along the route, and efficiency of the route. A further consideration is the reliability of the journey time. In Chapter 1, the literature also finds reliability to be an important aspect of service quality, which ultimately affects journey time.

Proximity of mode to origin/destination:

This indicator is part of the immediacy component of travel time. It attempts to measure how close the nearest node in the transport network is, both at the point of origin, and relative to the ultimate journey destination. This indicator is arguably most appropriate for comparing different modes – it is possible to generate a semi-qualitative rating according to the average number of nodes in the network, in the manner shown in Table 18.

Nodes in network	**Rating**	**Typically modal distribution**
5 or more per km^2	5	Cars, taxis, motorbikes, bicycles, pedestrians
Between 1 and 5 per km^2	4	Cars in parking restricted areas, buses
Between 0.1 and 1 per km^2	3	Buses
Between 0.01 and 0.1 per km^2	2	Trains
Fewer than 0.01 per km^2	1	Airports

Table 18 – Example inter-modal rating for proximity to origin/ destination, based on number of nodes per km^2.

The UK road network displays a high level of connectivity – as such, the number of nodes is very high, and most destinations are accessible by car. However, car access is restricted in urban areas, not by the road network, but by parking restrictions, and traffic management schemes such as congestions charging. Two proxy indicators are suggested, '*average cost of two hours of parking in urban areas*', and '*incidence of Penalty Charge Notices*' (parking fines issued by local authorities) which both reflect a combination of the scarcity of parking spaces in cities, and councils' desire to reduce traffic within them. For cost of parking, a press release from Direct Line Insurance (Direct Line, 2007) suggests that between 2000 and 2006 the average cost of two hours parking increased by 25%. Although it does not cover the full study period, this number could be used as a measure of the change of car access to urban areas between the two study years. However, it should be noted that an examination of the press release raises some questions as to the validity of the data – the percentage changes for individual areas included in article appear to have errors in their calculation. More reliable data were produced by the DfT relating to the issue of Penalty Charge Notices (PCN) between 2000 and 2005 (DfT, 2010b) by local authorities with Civil Parking Enforcement (CPE) powers. These are shown in Table 19. For the

London area, which has had these powers since 2000, numbers are given in thousands of PCNs issued. However, outside London an increasing number of local authorities received these powers across the time period considered. Therefore it is more appropriate to display the numbers as a ratio of PCNs to vehicles registered at addresses covered by CPE powers. Even so, these numbers do not provide an accurate description of change in infraction rates over time, as authorities with fewer parking issues may have been added, dragging down the average ratio. The most reliable are therefore the numbers for London, which show a clear rise in infractions.

	2000	**2005**	**Change**	
London	4,021	5,060	25.8%	Thousands of PCNs
Metropolitan areas outside London	0.60	0.25	-58.9%	Ratio of PCNs to vehicles registered at addresses covered by CPE powers
Non-metropolitan areas	0.35	0.24	-32.0%	

Table 19 - Incidences of Penalty Charge Notices issues by local authorities with Civil Parking Enforcement powers, 2000 and 2005, DfT (2010b).

Frequency of departure:

Frequency of departure is another aspect of the immediacy component of travel time. It is assumed that areas of service quality where the user is given a high degree of choice or control are deemed to be at ideal levels. Frequency of departure is one such area, at least from the perspective of the driver. However, it is impossible to quantify the frequency of departure from a passenger perspective, as this is entirely dependent on individual driver's availability. Given that it is assumed to be at 'ideal' levels, this indicator is deemed to be unchanged over the time period. It is another example of an indicator better suited to cross modal comparison, or analysis of public transport, where changes in frequency of services greatly affect the service quality.

Average speed:

Due to the wide variation in speeds depending on trip time and route, an overall average speed for the UK can only be an approximate measure. There are several sources of data. The first is the national travel survey by the DfT (DfT, 2006d and ESDS 2010a,b) which provides an average of total time spent travelling by car and total distance, from which an approximate speed can be calculated. This comes out to be 25.7 mph for 1995 and 24.4 mph for 2005, a drop of approximately 5%. Secondly, Transport for London (TfL) produces average speeds for traffic in London and the Greater London Area (TfL, 2006), which are presented in Table 20 (numbers were not available that exactly matched the study period – only the multiple year averages shown in the table were provided). These also show a drop of approximately 5%. Interestingly, the report notes that traffic speeds have gone up since the introduction of the London congestion charge in 2003, alongside a drop in traffic volume. This reinforces the point that the high demand placed on the road network is causing deterioration in service quality.

Mph	**1990-1997**	**2003-2006**	**Change**
Morning peak	15.7	14.8	-5.7%
Daytime	19.2	18.3	-4.7%
Evening peak	16.8	16.0	-4.8%

Table 20 – Average speeds in Greater London Area , mph, TfL (2006).

The DfT also produces statistics for speeds on UK roads (DfT 2006e and DfT 2007b). The statistics are split between data for speeds on roads comprising the strategic road network (SRN), (inter-urban or trunk roads) and data for urban roads gathered in major cities. The data are unfortunately not available for the entire time period considered, due to periodic changes in the methods used to gather data. For speeds on the SRN, presented in Table 21, the data are collected using vehicles termed ‘floating cars’. These are instrumented cars that drive along selected routes at the median speed, as a means of measuring traffic conditions. Since 2005, the speeds are based on Highways Agency's Traffic Information System (HATRIS) database, which includes journey time and traffic flow data. These are not directly comparable with the older data, but the 2005 values are nevertheless presented.

Mph	**1995**	**2003**	**Change**	**2005 (HATRIS)**
Morning peak	52.7	50.2	-4.7%	51.9
Daytime	55.4	55.2	-0.4%	55.6
Evening peak	54.5	51.3	-5.9%	52.2

Table 21 – average speeds on Strategic Road Network (Trunk roads), mph, DfT (2006e) and DfT (2007b).

In the case of speeds on urban roads (outside the London area), presented in Table 22, the data are based on surveys. Data for 1999-2000 were again not directly comparable with the numbers for 2000-2005 due to methodological differences. The number for 2005 in Table 22 is an average of 2004 and 2006, although there was no great change over the period.

Mph	**1999/00**	**2005 (interpolated)**	**Change**
Peak	21.8	20.95	-3.9%
Off-peak	26.3	24.65	-6.3%

Table 22 - average speeds in urban areas outside London, based on survey data, mph, DfT (2006e) and DfT (2007b).

With the exception of daytime speeds on the SRN, the changes in speed recorded by the DfT over the time period considered also cluster around a drop of 5%.

Reliability:

The reliability of car transport spans two issues. One is the reliability of the vehicles, and the likelihood of encountering mechanical failure. The other is the amount of variability in journey times due to changing conditions on the road network. The latter issue is separate from the simple question of speed: what is important is the level of predictability in the journey times. Commonly, people budget a certain amount of time to perform specific journeys. The greater the uncertainty surrounding the amount of time a trip will take, the larger the safety margin that needs to be build into the schedule. In this regard, congestion is not necessarily a problem, if it is predictable and consistent. However, the Eddington study (DfT, 2006c) notes that there is a strong positive correlation between congestion (a measure of average congested vehicle speeds against free-flow speeds) and travel time variability, with travel time predictability reducing as congestion increases. Therefore, the DfT measures reliability on the SRN with the average delay per 10 vehicle miles on the slowest 10% of journeys. This is not a perfect measure of variability – a better measure would be the proportion of days within a given percentage of the daily average speed along a route split by the time of day (morning peak, daytime and evening peak). These data are not available, but it is suggested that such a measure would better reflect the reliability of trip times. For urban areas, the DfT simply measures congestion using average minutes per mile during morning peak hours on selected urban routes in each region. In both cases (urban and inter-urban), the DfT has only started collecting data in the manner described since 2005, and is still developing its indicators. Therefore a comparison over the study period has not been possible.

	2005	
Inter-Urban trunk roads (SRN)	3.78	Delay per 10 vehicle miles (slowest 10% of journeys)
	1.16	Delay per 10 vehicle miles (all journeys)
Urban roads	3.50	Vehicle journey time – minutes per mile

Table 23 - Reliability indicators for 2005, minutes of delay, DfT (2006e) and DfT (2007b).

As mentioned above, the other issue concerning reliability is the likelihood of vehicle failure. Vehicle reliability data were taken from ADAC (the German automobile club) breakdown statistics for five specific vehicle models, selected on the basis of their popularity in the UK over the study period. These vehicles were consistently in the top 10 bestselling vehicles in the UK. The results are presented in Table 24 (1996 was used instead of 1995 due to data availability). It is clear that reliability has decreased over the time period across the models considered. An examination of the full dataset provided by

ADAC (1997, 2006) showed that this trend was not just restricted to the five models under scrutiny – a consistent decrease in reliability was observed. A large proportion (35%) of the failures in 2005 were electrical, a result of the increasing quantity of electrical components in vehicles, and the reliance on electronic management systems.

	1996	2005	Change
Ford Fiesta	18.73	13.80	-26.33%
Ford Mondeo	25.30	20.57	-18.71%
Renault Clio	20.00	41.10	+105.50%
Vauxhall Astra	21.23	29.30	+37.99%
Vauxhall Corsa	17.60	22.20	+26.14%

Table 24 - breakdowns per 1000 vehicles (average of 4-6 year old vehicles, adapted from ADAC 1997 and ADAC 2006)

4 .2.2. How comfortable is the trip?

Comfort is a difficult area to assess in a consistent and objective manner. Which aspects matter, and to what extent, depend on user preference. With this in mind, the approach has been to focus on the levels of choice or control the user has over his or her environment. It should be noted that there is a great deal of diversity in the car market, with a wide range of options on offer. In this, the car is unique in terms of the level of choice offered to users. However this is somewhat dependent on price.

Some elements of comfort suggested in Chapter 2, such as '*level of physical or mental effort*', '*exposure*' and '*privacy*' are better suited to cross modal comparisons as they have not changed. Others are entirely under the users' control and are thus deemed to be at ideal levels – these include '*access to services*' (insofar as the user can decide his or her route) and '*cleanliness*'. For the remaining aspects, the approach used to assess changes over the study period was to focus on changes made to the five models discussed in the previous, with data coming principally from 1995 and 2005 issues of 'What Car?' magazine (Payton, 1995 and Motton, 2005) This gives an idea of the change in overall standards.

Temperature:

The metric chosen to describe user control over temperature was the availability of air conditioning across each model range (each model is available in a range of variants, with different engines and equipment). This was available as standard on average on 18.3% of the range (all models) in 1995, and on 33.3% of the range (all models) as an optional extra, compared to 68.2% and 88.4% respectively in 2005. This suggests a greater prevalence of climate control in 2005 vehicles. Figure 10 provides a more detailed look at the changes.

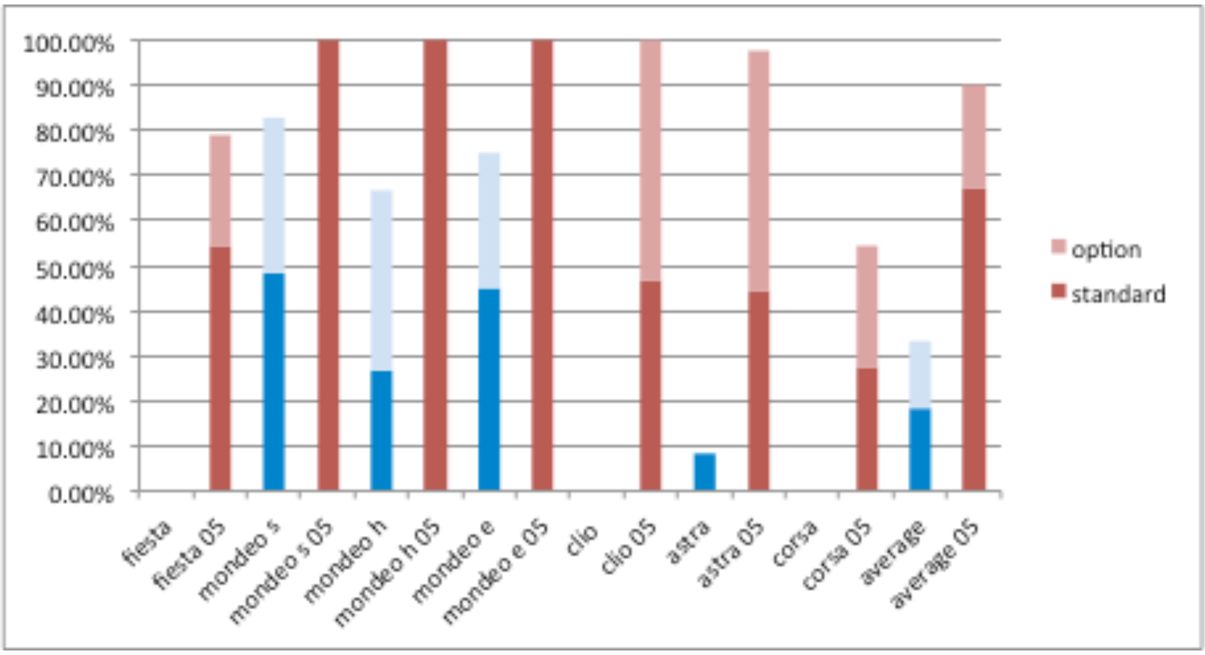

Figure 10 - Proportion of each model range with air conditioning (the three Mondeo types are the hatchback, saloon and estate versions). Adapted from Payton (1995) and Motton (2005). 1995 vehicles are represented in blue, 2005 in red.

Crowding/Space:

Detailed data for the cabin space available across the different models was not available. However, the overall size of the vehicles increased over the time period. It should be noted that an increase in overall size does not necessarily correspond to an increase in cabin space due to the increased amount of safety features in newer vehicles. Nevertheless, it is the best measure available. Average height across the five models increased by 4.4%, width by 5.7% and length by 4.9% over the time period. Chapter 3 discussed the increase in weight across the time period, which was of 14.5%. Although it is impossible to directly link increases in size with increases in weight, a size increase of some magnitude would be expected.

An alternative measure for space can be calculated by subtracting the average cargo space from the average cargo space with the rear seats down, as this provides some information on the amount of space available in the rear cabin of the vehicles. The difference is of 817 litres in 1995 and 855 litres in 2005, an increase of 4.7%. This suggests that vehicle cabin space is increasing.

Luggage capacity:

Cargo capacity has also increased over the time period. In 1995, the average boot capacity for the selected models was 368 litres and 1185 litres with the rear seats down. In 2005 this increased to 378 and 1233 litres, an increase of 2.5% and 4 % respectively. These increases are commensurate with increases in overall size.

Seat availability/quality:

This criterion is another indicator that is arguably better suited to intermodal comparisons. However, some data were available on some comfort related features available on vehicle seats. The metrics used are the availability of seats with manual height adjustment, electrically adjusted seats and availability of leather upholstery across the selected model ranges.

Year	1995	2005	Change		1995	2005	Change
	Standard equipment				Including availability as options		
Manual Adjustment	62.19%	76.67%	23.28%		69.46%	83.94%	20.84%
Electric Seats	11.56%	9.07%	-21.52%		12.50%	9.63%	-22.96%
Leather Upholstery	4.55%	10.25%	125.15%		12.54%	25.71%	104.92%

Table 25 - Seat comfort: percentage of model ranges with various comfort features

Table 25 gives the average availability of these features across the five model ranges previously discussed. It shows that although electric seats are available over a smaller percentage of the model ranges, overall the seat adjustability has increased, as has the availability of leather upholstery, suggesting that comfort levels are increasing gradually.

Price:

As a means of checking that improvements in the vehicle ranges are not just due to a greater variety of premium variants in 2005, a similar analysis was carried on prices. With the 1995 prices adjusted for inflation using the Retail Price Index (RPI)[5], the average cost across the selected model ranges dropped by 6.7%. Any changes in comfort are therefore not due to an increase in cost.

4 .2.3. *How safe is the trip?*

Safety has been an area of growing importance, with manufacturers making significant efforts to improve the safety of car users. This is shown through two sets of data. Firstly, Table 26 shows the European New Car Assessment Program (NCAP, 2011) crash test ratings for the selected models. These were instituted in 1997, but the 1997 tests were performed on essentially the same generation of vehicles as were available in 1995. The NCAP ratings consists of a five star rating system (five stars being awarded to the safest vehicles), which is assessed on the basis of performance in a series of tests including

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[5] VAT rate was also checked, and has remained constant during the study period.

frontal impact, car to car side impact, pole side impact, child protection, and whiplash. The selected models show an improvement from an average of 2.7 stars in 1997 to 4.4 in 2005. This increase in vehicle safety comes at a certain price however, in terms of an increase in weight, which has impacts on energy and material intensity, as well as emissions. Chapter 3 notes that around 27% of the increase in average vehicle weight during the study period is attributable to additional safety features.

	1995	**2005**	**Change**
Ford Fiesta	3	4	33%
Ford Mondeo	2.5	4	60%
Renault Clio	2	5	150%
Vauxhall Astra	4	5	25%
Vauxhall Corsa	2	4	100%
Average	2.7	4.4	63%

Table 26 - NCAP safety ratings for selected models, stars out of five, (NCAP 2011).

The second source of data is the accident statistics. These show a reduction in car user casualties per passenger-km for all injury types (Table 27). Interestingly, the smallest improvement is found in the number of fatalities, suggesting that the greatest improvements are being made at lower speed collisions. These statistics confirm that driving is becoming safer – however, it is worth noting that the statistics do not directly validate the NCAP ratings, as the vast majority of vehicles on the roads in the respective years were not manufactured in those years. The NCAP ratings only begin to become applicable to accident statistics from approximately 2007 onwards, by which time over 75% of vehicles in the fleet will have been subject to the testing process. Further examination of impacts related to road accidents is performed in Chapters 5 and 6.

	1995	2005	Change
Casualties	32.1	25.4	-20.9%
Of which: killed	0.29	0.24	-17.9%
Seriously injured	3.63	1.85	-48.9%

Table 27 - Casualties per 100 million passenger-km, DfT (2006a).

A final concern is the security of the users with respect to crime. Dealing with this requires a different approach to that of user safety from accidents. This is because the risk of crime exists independently of the transport, whereas the risk of accidents is directly linked to the transport itself. Therefore, it makes more sense to focus on a modes' ability to protect from crime, than to quantify the incidence of it. The car fleet, by virtue of being private transportation, offers good protection against assault and robbery. Obviously this has not changed over the time period. Were a cross modal comparison made, a qualitative measure could be used or, alternatively, a measure of incidents per passenger per year relative to the national average of incidents per person per year, as a means of determining the relative security of the mode in question.

Ownership of a car does however open the user up to the risk of theft. It is likely, given developments in anti-theft technology, that this risk has reduced over the time period. However, no statistics were found, as the police statistics classify vehicle theft under the heading of property crime, and do not record the theft of vehicles specifically.

4 .2.4. Equality of Modal Opportunity

A transport system needs to be assessed not only on the quality of service delivered to its users, but on how accessible[6] it is to the community and whether any groups within the

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[6] The section is called 'Equality of Modal Opportunity', where 'Access' or 'Accessibility' could have been used instead. However, in some transport literature, 'accessibility' is defined as 'access to goods and services' rather than 'access to a mode', as in Banister (2007) The term "equality of modal opportunity" was chosen to avoid confusion.

community are excluded from using it. An ideal transport system would be able to meet the transport needs of the entire community. If a mode is not accessible to the entire community, a portion of the community is excluded from using it, and must meet its transport needs using other modes.

Affordability:

A key requirement for equality of modal opportunity is affordability, the extent to which socioeconomic status affects access to and use of the mode. Table 28 shows the percentage of households with access to one or more private and light goods vehicles, split by income brackets. This is the lowest criterion for access as it does not differentiate based on driver or passenger status, or the number of vehicles the household has access to.

The data are from the National Travel Survey (NTS) by the DfT, and are available both in the form of reports (DfT, 2006d) and datasets accessed through the Economic and Social Data Service, a UK data archive (ESDS 2010a,b). The 1995 numbers are actually based on data for 1995, 1996 and 1997 due to the sample size being insufficient for the analysis of a single year. Since 2002, the data collected in the survey are sufficiently detailed to seperate households along income quintiles. Income quintiles, by definition, contain 20% of the population, and are thus different to income brackets, which split the population according to specific income thresholds. Unfortunately, this level of detail was unavailable for 1995, so the results in Table 28 use income brackets. It can be seen from the results why income quintiles are more useful than income brackets – from looking at the results per bracket, it would appear that access to cars has reduced over the study period. However the ‘all households’ category shows that overall access has increased. This is because the proportion of households in the three income brackets has changed. For instance, the lowest income bracket contained 71.1% of households in 1995, compared to just 52.7% in

2005. The middle bracket went from 23.7% of households to 31.2%. This means that overall affordability has actually increased.

1995 income categories	1995/1997	2005
Less than £25000	59.2%	59.9%
£25000- £50000 (not inclusive)	95.2%	90.6%
£50000 or more	95.3%	93.7%
All households	69.6%	74.9%

Table 28 - Households with access to one or more cars, split by income brackets

It is possible to calculate an approximation of the 1995 'low income' bracket, based on the fact that this bracket contains the lowest 71% of households by income. Using the 2005 income quintile data, it is possible to calculate an estimate of access to cars for the lowest 70% of households by income for 2005. This works out to be approximately 69%, indicating a 10% increase in lower income households that have access to a car. This conversion is not exact, but taken alongside the overall increase in access to cars (70% to 75%) it is clear that access based on income has become less of an issue over the study period. Lucas and Jones (2009) confirm this in their report 'The Car in British Society', noting also that town population size has become a better indicator of car access than income, with residents of rural areas having a greater access to cars than residents of large cities that offer viable alternatives. This suggests that cost is no longer the limiting factor for car access as much as choice and viability of alternatives.

A preferable indicator would be a ratio of the proportion of households with access to a car in the highest income quintile relative to the lowest income quintile. This would give a better sense of the extent to which income disparity affects access to private vehicles. This is dependent upon data being available in terms of income quintiles, which appears to have become the norm in the National Travel Survey.

Age, Infirmity and Disability:

Age, infirmity and disability affect the level of access to the mode, and the service quality experienced whilst using it. A disabled or infirm individual will have different requirements in terms of comfort for instance, but may also have a different experience in terms of the proximity they can achieve to their destination, due to disabled parking spaces.

Access as a driver is determined by the possession of a driving license. In the UK, the minimum age to obtain a driving licence is 17, and this is valid until the age of 70. Beyond that age, the licence must be renewed every three years, and each application requires a medical self-declaration by the applicant. Table 29 shows the proportion of the population in possession of a driving licence, split by age group. Interestingly, although overall licence possession and licence possession amongst the elderly has gone up, there are fewer licence holders in the younger age brackets. This could be linked to higher university attendance reducing the need for, and affordability of cars, although this is just speculation. The increase in driving licence possession overall and in the elderly is predominantly due to an increase in women drivers (the overall female licence ownership went from 57% to 63%, whereas the overall ownership for men remained unchanged).

	all aged 17+	**17-20**	**21-29**	**30-39**	**40-49**	**50-59**	**60-69**	**70+**
1995/1997	69	43	74	81	81	75	63	38
2005	72	32	66	82	84	82	74	51

Table 29 - Percentage of Driving Licence holders by age, from DfT (2007a).

With regards to disability and infirmity, the DVLA issues a guide on fitness to drive (DVLA 2011), detailing the medical requirements for drivers. Disability does not necessarily exclude an individual from driving, although in some cases the vehicle controls may need to be modified. Regardless of whether the infirm or disabled individual is a driver, from a comfort perspective the car fleet is unique in terms of how much choice is available to users, and how closely vehicles can be tailored to their needs. There are various sources of government support for mobility, such as the Specialised Vehicles Fund for people requiring complex adaptations. Quantifying the benefit provided by the car fleet to disabled and infirm citizens is difficult, due to the wide variety of severity of disability or infirmity. The metric selected was the number of parking badges for disabled people (blue badges) on issue in England (UK wide figure were not available). The numbers are shown in Table 30, both on a total and a per capita basis. The per capita basis is included to show that adoption of blue badges did not simply increase as a function of population growth. These badges grant people with a disability access to more conveniently located parking spaces, and as such improve their driving experience. They are also a useful indicator of the number of disabled or infirm people using the car fleet. It is apparent from the data that this number has increased substantially, suggesting that a greater number of disabled people are now using the car fleet, or are at least experiencing a higher level of service quality (in terms of parking convenience) whilst doing so.

	1995	2005	Change
Number of blue badges on issue	1463	2119	+44.84%
Number per 1000 population	30	42	+40.00%

Table 30 - Number of blue badges on issue (thousands/number per thousand) in England

It should be noted that if access to cars is limited to being a passenger, this affects the overall service quality of the car. For instance, the frequency of departure can no longer be deemed quite as ideal, as it is dependent on the driver, as are other aspects such as access to services (driver chooses the route, and when to stop), privacy, and to some extent destination.

4 .3. Employment

Employment provides a positive social impact; it allows people to support themselves and their families. For the purpose of this study mainly the quantity of jobs is considered. It can be argued that the quality of jobs provided (in terms of the skill level required, training provided etc.) is as (or more) important than the quantity of jobs. Whilst there is merit to this argument, in the majority of cases any employment is better than none. Some recognition of job quality is provided by the inclusion of costs to business per employee. An increase in this figure is assumed to yield greater benefit to the employees.

The industry surrounding the manufacture, sale and maintenance of motor vehicles is sizeable – it represents approximately 2.75% of jobs in the UK in 2005. Table 31 shows the average employment during each year, as well as the costs (to businesses) per employee, which cover gross wages and salaries (including redundancy and severance payments) and social security costs. The data were taken from the Annual Business Inquiry, published by the Office for National Statistics (ONS, 2007). The data covered retail, manufacture and maintenance of all motor vehicles and fuel indiscriminately so a correction was made to estimate the contribution of private and light goods vehicles. The correction was broadly made according to the proportion of private and light goods vehicles produced by UK manufacturers. A more precise explanation of the correction is described in section 6 .1. Further to this correction, the data for employment did not reach back to 1995, so this was extrapolated by 3 years. Figure 11 shows the employment in ‘sales and repair’ (excluding manufacturing) based on the available data, which illustrates that such an extrapolation is reasonable.

	1995		2005		Change	
	Employment (thousands)	Costs/ Employee (£)	Employment (thousands)	Costs/ Employee (£)	Employ-ment	Costs/ Employee
Manufacturing	282*	25954	205	31617	-27.3%	21.8%
Sales (vehicles, parts, fuel), and repair	562*	11965	560	17383	-0.4%	45.3%
Total	843*	16640	765	21195	-9.3%	27.4%
*Denotes extrapolated data						

Table 31 - Employment Average and Employment Costs per Employee in Automotive Industry, adapted from ONS (2007).

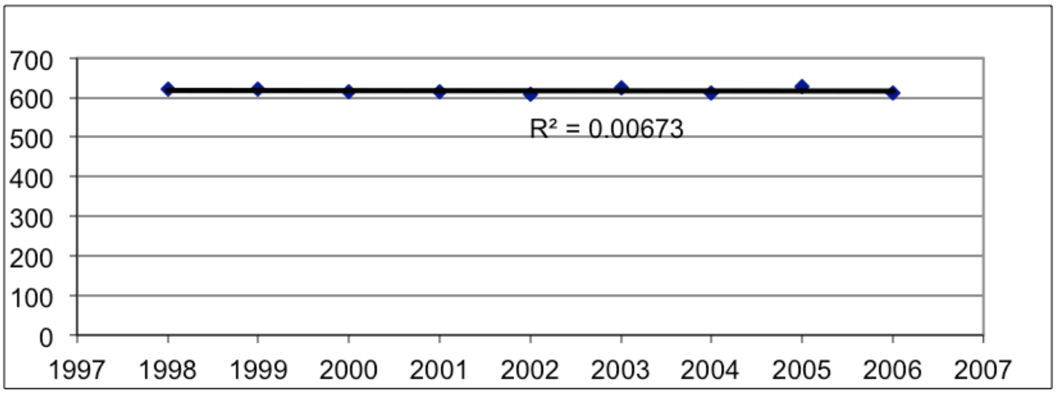

Figure 11 - Employment (average during year) in Sale, Maintenance and Repair of motor vehicles and motorcycles; retail sale of automotive fuel, thousands per year, ONS (2007).

The insurance industry also contributes to employment, although no appropriate data were found to describe this impact. Overall, there has been a drop in employment over the time period. The biggest decline is in the manufacturing area, but there was also a small reduction in the retail (vehicles, parts and fuel) and repair sector. Some unpacking is required for this sector. It is comprised of several subcomponents (retail of vehicles, retail of parts, retail of fuel and maintenance). All of these, with the exception of maintenance actually saw a reduction in employment numbers. However, the maintenance sector grew considerably, by almost 40%. This is understandable given the increase in fleet size of approximately 28%. However, it is less clear why employment in other sectors declined, despite increases in fuel consumption and annual new car registrations. It appears likely (given increases in value added – see section 6 .1.) that these industries have simply become more efficient in terms of employees required.

4 .4. Summary and Conclusions

Overall, it is fair to say that the impact on the Human/Social store of value is mixed. Firstly, it is clear that more mobility is being provided, which in the context of this study is a positive impact. However, the picture is less clear when considering the quality of the mobility provided. In terms of what could be considered the key performance characteristics of a journey, namely speed, reliability and convenience, quality has dropped. On the other hand, other characteristics such as comfort, safety and affordability, have seen significant improvements over the period. It should be noted that although aspects such as comfort may appear frivolous, they do have wide ranging sustainability impacts. Air-conditioning for instance affects fuel efficiency, as does vehicle weight, which is increased not only by better equipment and more spacious cars, but also by improved safety features. Speed also has wider ranging effects: congestion increases fuel consumption, whilst exceeding the speed limits (although not explicitly considered here) affects safety and increases fuel use. More importantly, users deem these aspects important. The car fleet, due to its decentralised nature, is ultimately driven by user decisions, so these aspects must be considered to understand the sustainability of the system and how to change it.

Chapter 5

Health Impacts of the UK car fleet

5 .1. Impact of Air Emissions on Health

The contribution of the car fleet to air emissions, discussed in chapter 3, impacts more than just the environmental store of value. There is a body of evidence linking air pollution to impacts on human health affecting the human/social store of value. In their review of recent research into the health effects of air pollution, Ren and Tong (2008) conclude that air pollution causes consistent short-term effects on hospital admissions and deaths and that there is also evidence linking air pollution to long-term effects. Epidemiological studies of the impacts of particulates on human health have been reviewed by Pope (2007), and for various pollutants by Sahsuvaroglu and Jerrett (2007). In the UK, the Committee on the Medical Effects of Air Pollution (COMEAP) have published a series of reports attempting to assess the impacts of air pollution in the UK (COMEAP 1998, 2010) and reviewing existing studies to determine the most appropriate means of doing so (COMEAP 2001, 2009). They identify $PM_{2.5}$ as the pollutant most strongly associated with increased risks of mortality.

The indicators (Table 32) chosen to describe this impact are 'deaths brought forward due to car-related air pollution' (representing the mortality impact) and 'hospital admissions caused by car-related air pollution' (representing the morbidity impact), in the given year. There are four key pieces of data to quantify the impact of air pollution on 'deaths brought forward':

1. The ambient levels of pollutants;

2. An estimate of the share of the impacts attributable to the car fleet;
3. Exposure-response coefficients (ERC) for mortality and morbidity;
4. Mortality data.

Only direct (tailpipe and tyre and brake wear) vehicle emissions are considered in this section as these have the clearest impact on ambient air quality in populated areas.

Impact	Issue	EIR	Indicator	Metric
Air Pollution	Air pollution causes accelerated death	Communities	Deaths brought forward due to car-related air pollution	#
Air Pollution	Air pollution causes respiratory disease	Communities	Hospital Admissions due to car-related air pollution	#
Air Pollution	Air pollution causes accelerated death and respiratory disease, which places a greater burden on the health services	Communities	Cost to health services of emissions related hospital admissions	£

Table 32 - Human/social health impacts related to emissions of the UK car fleet.

5 .1.1. UK Air quality

UK measurements are collected from monitoring stations at nine different types of locations (AEA 2010a). This study focuses on urban and suburban areas, as these areas are the most heavily populated and are also the most affected by vehicle emissions. The annual means from urban, suburban and roadside sites were therefore averaged to produce overall estimates for the UK. A more accurate method would be to overlay air quality data for the whole country with census data, to produce a population weighted average. This was the approach used by COMEAP (2010) in which they consider air pollution in 2008. However, this approach wasn't feasible within the constraints of this study. An estimate was made using the approach used in this study for 2008 in order to make a comparison with the COMEAP (2010) study. The resulting estimate of mean $PM_{2.5}$ was approximately 9% higher than the COMEAP estimate for population-weighted mean $PM_{2.5}$ in the UK, which was expected given that our estimate related to urban and suburban areas, and not the UK as a whole.

The data (Table 33) are consistent with overall trends in emissions. The only increase over the period is in the concentration of ozone, due in part to decreases in NO_x emissions (at high NO_x concentrations, O_3 becomes scarce due to NO oxidising to NO_2,) (Mattaj & Hutchinson, 2005). Measurements of $PM_{2.5}$ were not available for 1995, and were only available for 4 sites in 2005. Therefore, the $PM_{2.5}$ concentration was estimated by applying a single ratio (from DEFRA, 2005) to the concentration of PM_{10}. This method of estimating levels of $PM_{2.5}$ was also used by Beelen *et al* (2008). From this was subtracted a value of 1.42µg / m^3, the amount of air pollution attributed to non-anthropogenic sources by COMEAP (2010). This was assumed to stay constant over the study period.

	1995	**2005**
PM_{10}	29.81	23.38
$PM_{2.5}$ (Based on single ratio adjustment of PM_{10})	16.90	13.26
$PM_{2.5}$ (Anthropogenic fraction)	11.86	17.13
SO_2	25.11	4.77
NO_2	50.50	37.43
CO	0.85	0.37
Ozone (Annual mean of Daily Max 8 Hour Mean)	53.20	56.15

Table 33 - Annual means of ambient levels of air pollutants, µg / m^3, calculated with air quality data from AEA (2010a).

The car fleet is not solely responsible for ambient air pollution; however attributing the share of ambient levels of air pollution it is responsible for is difficult, especially for particulates. DEFRA (2005) notes that there are many natural sources of particles, which contribute significantly to ambient concentrations, and that these particles tend to be difficult to predict in terms of their magnitudes and emission rates . In a similar vein, the London Atmospheric Emissions Inventory report (LAEI, Mattai and Hutchinson, 2008) notes that:

> *"Many of the processes that lead to PM10 in the atmosphere are difficult to describe in a deterministic way e.g. wind-blown dusts. Furthermore, a significant proportion of measured PM10 concentrations are derived from the oxidation of species in the atmosphere such as NOX and sulphur dioxide. These processes can take place over hundreds to thousands of kilometres and therefore involve other UK and European sources."*

As a result, source apportionment is at best uncertain, even if sophisticated models are used.

In this study, the share of annual emissions was used as a basis for attributing the health impact due to the car fleet. Table 34 displays the proportion of national emissions produced by private and light goods vehicles (DEFRA, 2005). It can be seen that for most pollutants the share of pollution caused by the the private and light goods fleet has been shrinking despite significant reductions national all source emissions. This means that the car fleet has been reducing emissions faster than the national average. The one exception is in the emissions of PM_{10}, where despite the fact that emissions from the car fleet have reduced, the share of national emissions has actually increased, indicating that the car fleet is somewhat lagging behind the national average in this area.

		PM_{10}	$PM_{2.5}$	SO_2	NO_x	CO
Percentage change in national, all source emissions – 1995 to 2005		**-37.0%**	**-**	**-69.6%**	**-31.8%**	**-61.6%**
National Share – car fleet	1995	12.5%	-	1.2%	29.3%	63.1%
	2005	16.0%	-	0.3%	16.8%	41.6%
	Percentage Change	**+3.5%**	**-**	**-0.9%**	**-12.5%**	**-21.5%**

Table 34 – UK fleet's share of national emissions.

Despite the issues with modelling particulates, COMEAP (2010) were able to apportion $PM_{2.5}$ in the UK for 2008, using a model described in Stedman et al. (2007). Their results for local traffic are shown in Table 35 as a basis for comparison. Their estimate of a 6.8% share of particulates across the UK ($PM_{2.5}$ in this instance, although PM_{10} and $PM_{2.5}$ are very highly correlated) is much lower than the estimate of 16% in Table 34, but their estimates for urban areas are closer.

$PM_{2.5}$	Inner London	Outer London	UK
Local Traffic	13.2%	9.7%	6.8%

Table 35 - Source apportionment results for $PM_{2.5}$ in the UK, 2008, taken from COMEAP (2010).

It should also be noted that the emissions share in Table 34is for NO_x emissions rather than NO_2. NO_x is comprised of both NO and NO_2, so these are not analogous. However, relative concentrations of NO and NO2 are difficult to determine as they are influenced by the concentration of ozone. No share is given for ozone because it is not directly emitted, but is dependent on climate as well as the presence of precursor emissions, namely NO_x, Non-Methane VOCs, CO and Methane (AEA 2010b).

5 .1.2. *Impact on Mortality*

Calculating the mortality burden of air pollution for a specific year presents a number of interesting problems, due to the fact that mortality rates in the current year are affected by air pollution in past years as well, and that levels of air pollution in the current year will affect mortality rates in subsequent years. The method used is to compare two scenarios for the year under scrutiny – a baseline case with observed death rates, and an alternative case where these death rates are reduced by an amount based on the levels of air pollution that year and the corresponding exposure-response coefficient[7] (ERC). The result produced this way represents the effect of air pollution at a population wide level, but is not applicable at an individual level. This means that it does not equate to the number of individuals in whose deaths air pollution played a part – this number is probably greater, but to a lesser

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[7] ERC's are expressed in terms of a percentage increase in an outcome per increment increase in pollutant levels. They are presented with high, low and mean estimates, representing a 95% confidence interval. In situations when the associations between exposure to a pollutant and a particular event are not clear, and comprise more uncertainty, it is possible for ERC's (or at least the low estimate) to be negative.

degree. It is also incorrect to use this result in the context of considering 'deaths avoided' or 'lives saved' by reducing air pollution. Over time, a reduction in air pollution would alter the population size and age structure, ageing it, and total mortality would settle back to what it was before. COMEAP (2010) suggest it is most appropriate to describe the results as "*an effect on mortality equivalent to 'X' deaths*".

The calculation for evaluating the impacts of air pollutants (from all sources) was adapted from Sahsuvaroglu and Jerrett (2007), but is effectively the same as the one used in COMEAP (2010):

$$\text{Deaths Brought Forward} = \text{Deaths}_{\text{year, n}} - \frac{\text{Deaths}_{\text{year, n}}}{(\text{ERC} \times \text{Pollutant Concentration}) + 1}$$

This assumes a linear relationship between pollutant concentrations and health impact. Sahsuvaroglu and Jerrett (2007) also consider a baseline pollutant concentration below which no impact occurs, but this was not done here.

The formula can be derived in the following way:

$$if\ Deaths\ Brought\ Forward = Deaths_{\text{year,n}} - Deaths_{\text{year, n, no emissions}}$$

$$\begin{aligned} and\ Deaths_{\text{year,n}} \\ &= Deaths_{\text{year, n, no emissions}} \\ &+ Deaths_{\text{year, n, no emissions}} \times ERC \times Pollutant\ Concentration \end{aligned}$$

$$then\ Deaths_{\text{year,n}} = Deaths_{\text{year, n, no emissions}} \times (1 + ERC \times Pollutant\ Concentration)$$

$$and\ Deaths_{\text{year, n, no emissions}} = \frac{Deaths_{\text{year, n, no emissions}}}{(1 + ERC \times Pollutant\ Concentration)}$$

It should be noted that when using this formula to calculate *deaths brought forward* with the high and low estimates of an ERC, the results will not scale in quite the same way as the ERC's would suggest. For example, an ERC of 6% (and assuming a single increment pollutant concentration) would result in 5.7% of annual mortality being attributed to pollution, whereas an ERC of 12% would result in 10.7% of annual mortality being attributed to pollution, which is a little less than double.

The choice of exposure-response coefficient (ERC) is critical to the calculated health outcome. ERCs are derived from epidemiological studies, which measure the change in recorded mortality and morbidity due to an increase in a pollutant. There are two types of study: time-series studies for short-term responses (typically a few days), and cohort studies for long-term outcomes of exposure to a higher pollutant concentration. Sahsuvaroglu and Jerrett (2007) note two points about short-term studies. First, it is unreasonable to assume that short-term responses would be sustained given a permanent change to ambient pollutant concentrations. This is because a short, sudden pollution episode could worsen the state of already frail individuals unaccustomed to higher pollutant concentrations, bringing forward their death or disease by as little as a few days. Thus, in calculating longer-term impacts e.g. annual mortality attributable to air pollution, the outcome could be inflated. Secondly, only acute effects are captured, so that the outcome could be severely underestimated through the neglect of chronic impacts. Cohort studies capture the response to both long-term and short-term increases in exposure (COMEAP 2001, 2009). For this type of study, which aims to assess impacts over a year, cohort studies are most appropriate. However, cohort studies are not always available for the outcome and pollutant under scrutiny.

Although there are short-term associations between a number of pollutants and mortality, not all pollutants have been linked to long-term effects. COMEAP (2009, page 9) recommends $PM_{2.5}$ as the quantitative assessment index of all-cause mortality for policy interventions, adding that:

> *"Evidence relating to the possible effects of long-term exposure to the common air pollutant gases (sulphur dioxide, nitrogen dioxide and ozone) is*

> *less well developed and we do not make any recommendations in favour of quantifying the effects of long-term exposure to these compounds."*

For ozone, Jerret *et al.* (2009) detect no increase in all-cause mortality in a cohort study. Beelen *et al.* (2008) also fail to find significant associations with SO_2, although do find an association with NO_2. For the purposes of calculating the indicator in this study (*deaths brought forward*) $PM_{2.5}$ was therefore used as the sole marker. COMEAP (2009) recommends using a mean value of a 6% increase in mortality per $10\mu g/m^3$ of $PM_{2.5}$, with low and high values of 1% and 12%, respectively.

Mortality data were supplied by the UK Office for National Statistics (ONS, 2005 - Deaths: underlying cause, sex and age-group). The figure taken was for non-traumatic mortality (i.e. annual deaths minus injury and poisoning) in over 30's in England and Wales. Only over 30s were considered since the cohort studies on which the ERC's are based only applied to that age group. The data were restricted to England and Wales due to the way in which mortality statistics are recorded in the UK – Scotland and Northern Ireland Record their statistics separately, and although the headline figures are available, the age and underlying cause breakdown is not. Urban mortality was assumed to be 84.9% i.e. the proportion of people living in an urban environment (Hutchinson and Pearson, 2004).

By bringing together the ERC's, the air quality data, the estimate of the share attributable to the car fleet, as well as mortality data, an estimate can be made of '*deaths brought forward'* due to car emissions. The calculations, summarised in Table 36, suggest that the impact of air pollution is significant – total fatalities (due to collisions) amongst private and light goods *users* were 1729 in 2005 (DFT, 2006). Looking at the Mean ERC figures, *deaths brought forward* due to car emissions in 2005 dropped by 17.5% from 1995, despite the number of private and light goods vehicles increasing from 22m to 29m in the same period.

This is due to the significant drop in emissions by the private and light goods fleet, the result of increasingly stringent emissions standards (EC, 2007), which were summarised in Chapter 3, Table 7. The EURO 2 standards came into force in 1996, and EURO 3 in 2000. Given that manufacturers typically make changes to their vehicles ahead of the legislation being enforced, by 2005 these regulations will have significantly impacted the make up of the fleet.

		Low ERC	Mean ERC	High ERC
Total – air pollution from all sources	1995	7719	42717	78152
	2005	4857	27531	51633
	Percentage Change	-37%	-36%	-34%
Share – attributable to private and light goods vehicles	1995	965	5340	9769
	2005	777	4405	8261
	Percentage Change	-19.5%	-17.5%	-15.4%

Table 36 - Numbers of 'deaths brought forward' in England and Wales using long-term studies of $PM_{2.5}$.

'Deaths brought forward' do not capture the mortality burden fully unless the age at death is considered also. This is apparent in the above comparison with accident data, as the life-years lost per 'death brought forward' may be fewer than the life-years lost due to a fatal car accident. Using the mean ERC figures, it was possible to estimate the total number of years of life lost (YOLL) for each year. The same method as COMEAP (2010) was used, which makes use of the IOMLIFET spreadsheets (Miller and Hurley, 2006). The mortality risk at each age is calculated using mortality data broken down by age and population data. A factor is then calculated using the ERCs and air quality data to reduce the mortality risk at each age to what it would be given no anthropogenic air pollution. This new mortality risk is then multiplied by the original population data to calculate mortality given no air pollution. The difference in number of deaths between the two cases is then multiplied by the expected remaining life expectancy at each age (again assuming no anthropogenic air pollution) to give total life years lost. The results are given in Table 37. The YOLL per attributable death work out at approximately 11.7 for 1995 and 11.6 for 2005. It should be noted that this does not imply that in 2005, 4405 people died due to car air pollution who would have otherwise lived a further 11.7 years. It is possible, even likely, that a greater number of people lost fewer life years. This is simply a means of estimating the overall mortality impact.

Calculating the annual YOLL due to air pollution allows a more accurate comparison with accident statistics. Performing a similar calculation on annual road casualties (see section 5 .2) reveals that the years of life lost due to accident fatalities resulting from accidents in which private and lights goods vehicles were involved is approximately 113700 for 2005. This is roughly twice the estimated impact of car related air pollution, which is approximately 51052 years of life lost for 2005. This largely due to the fact that YOLL per road fatality is roughly 40 years, which is much higher than the YOLL per pollution attributable death. This shows that that the impact on health due to air pollution should not be overlooked, but that road safety remains the higher priority.

	1995	**2005**	**Change**
Total – air pollution from all sources	500337	319072	-36.2%
Share – attributable to private and light goods vehicles	62542	51052	-18.4%

Table 37 - Life years lost in England and Wales using the mean ERC from long-term studies of $PM_{2.5}$.

5 .1.3. Impact on Morbidity

The morbidity impact is calculated in much the same way as the mortality impact, based on numbers of hospital admissions for different primary diagnoses. However, the choice of exposure-response coefficient is complicated by the fact that evidence for effects on morbidity is more fragmented than for mortality. Most of it is based on time-series studies, although there are some cohort studies focusing on the impacts of chronic exposure.

In a review of European cohort studies focusing on the effects of air pollution, Brunekreef (2007) notes that there have been several studies linking air pollution (mainly NO2, PM2.5 and Black smoke) to incidences of allergy and asthma in children. He also notes that a Europe-wide cohort study found no such links with adults, although studies focusing on singles cities found links between NO2 and bronchitis symptoms. Gotschi *et al.* (2008) review evidence on the long-term effects of air pollution on lung function. They conclude that most studies show statistically significant adverse effects, broadly across two patterns. The first is reduced lung growth in children exposed to higher levels of air pollution, and the second is a correlation between lung function levels in adults and pollution exposure. There is evidence to suggest that air pollution leads to an acceleration of lung aging. Karakatsani *et al*. (2003) find evidence linking air pollution with chronic pulmonary disease. Schindler *et al*. (2009) use results from the SAPALDIA cohort study in Switzerland to link a reduction in PM10 with a reduction in a variety of respiratory symptoms such as coughing or wheezing. None of these studies however provide satisfactory ERC's linked to hospital admissions.

Some studies focus on links to cardiovascular (CV) morbidity. Gan *et al.* (2011) link long-term exposure to traffic-related air pollution to an increase in coronary heart disease (CHD) hospitalisation in addition to mortality. They found a linear exposure-response relationship for black carbon and CHD hospitalisation and mortality, but not for $PM_{2.5}$. NO_2 also showed a linear response, but this was mostly attenuated after adjustment for carbon black. An interquartile range elevation (approximately 0.8 $\mu g / m^3$) in black carbon was associated with a 3% increase in CHD hospitalisation. In England, Forbes *et al.* (2009) examine the effects of various air pollutants on doctor-diagnosed angina, heart attacks or strokes, as recorded by the Health Survey for England. They looked at 1994, 1998 and 2003. They found a weakly positive association between PM10 and cardiovascular disease, although the size of the effect was large: a 29% increase in cardiovascular disease per 10 $\mu g / m^3$ for men, and a 16% increase for women. Beelen *et al.* (2008) consider long-term exposure to traffic-related air pollution and the risk of lung cancer, but find only a small effect on non-smokers. In a similar vein, Gallus *et al.* (2008) find no clear association between PM10 and lung cancer.

There are many time-series studies examining air pollution and morbidity. Sahsuvaroglu and Jerret (2007) review a number of studies and use them to create estimated exposure-response coefficients for PM10, NO2, SO2, CO and O3. Dominici *et al.* (2006), and Bell *et al.* (2009) perform studies on over 9 million Medicare enrolees in the US, looking at PM2.5 and Carbon Monoxide respectively. They find associations with CO and cardiovascular disease, as well as PM2.5 and both respiratory and CV diseases. The ERCs produced are lower than those suggested by Sahsuvaroglu and Jerret, but in their study Dominici *et al.* (2006) note that there is a high level of heterogeneity in terms of magnitude of response across the country, based on the exact composition of the particulates and other factors such as temperature. This makes comparisons between different studies difficult. Glorennec and Monroux (2007) conduct a health impact assessment on the effects of PM_{10} in the city of Caen in France, based on ERCs produced by the European APHEA study. These are closer to those presented by Dominici *et al.*

Table 38Table 39 present results for hospital admissions due to car related air pollution based on the ERCs presented by Sahsuvaroglu and Jerret (2007). Data for hospital admissions was taken from HES (2006). It should be noted that ozone would be included, but is not due to difficulties in attributing the share. The “adjusted mean” is a corrected ERC included by Sahsuvaroglu and Jerret to allow for an error in the software used in many time-series studies, which they calculate could account for an overestimate of up to 42%. Finally, it should be noted that the totals are not strictly accurate – not only do they not include a contribution from ozone, but it is problematic to assume that the impacts of each pollutant are additive. Although multi-pollutant studies were preferred for deriving the ERCs, it is difficult to isolate the impacts of individual pollutants completely. Brunekreef (2008) notes that NO2 and particulates are often highly correlated.

	1995				**2005**			
Pollutant	Low	Mean	High	Adj. mean	Low	Mean	High	Adj. mean
PM10 ($\mu g/m^3$)	1548	4444	7103	2653	1516	4405	7119	2610
SO2 (ppb)	82	228	367	134	4	12	19	7
NO2(ppb)	4236	18837	31532	11479	1925	8764	14983	5276
CO (1 ppm)	-	-	-	-	-	-	-	-
O3 (ppb)	-	-	-	-	-	-	-	-
Total	5866	23509	39001	14266	3444	13180	22121	7894

Table 38 – 1995 and 2005 respiratory hospital admissions due to car related air pollution based on ERCs from time series studies (Sahsuvaroglu and Jerret, 2007, ERCs).

	1995				**2005**			
Pollutant	Low	Mean	High	Adj. mean	Low	Mean	High	Adj. mean
PM10 ($\mu g/m^3$)	1579	4298	6867	2503	1407	3858	6211	2235
SO2 (ppb)	18	98	186	54	1	5	9	2
NO2(ppb)	24299	34397	43550	21292	10269	14709	18825	8968
CO (1 ppm)	147	716	918	404	408	1978	2532	1119
O3 (ppb)	-	-	-	-	-	-	-	-
Total	26043	39509	51521	24253	12085	20550	27576	12324

Table 39 – 1995 and 2005 cardiovascular hospital admissions due to car related air pollution based on ERC's from time series studies (Sahsuvaroglu and Jerret, 2007, ERCs).

Results based on other ERCs are also presented. Firstly, the APHEA coefficients (Table 40), as these were based on European studies, and secondly the Forbes (2009) coefficients, as these were based on an English study, during a similar timeframe to this study, and which considered long-term effects. PM_{10} was used above the gaseous pollutants as it showed the clearest association with cardiovascular morbidity.

	1995			2005		
Pollutant: PM10	Low	Mid	Upper	Low	Mid	Upper
Resp. morbidity	1331	1977	2820	1302	1940	2776
Resp. morbidity >65	428	635	906	665	991	1419
Cardiac morbidity	638	1579	2503	567	1407	2235
Cardiac morbidity >65	755	1309	1852	819	1423	2018

Table 40 - Hospital admissions due to car related air pollution based on APHEA ERC's – note that the same coefficients were used for respiratory morbidity and respiratory morbidity in over 65 year olds. Also note that for 2005 the admissions data for those aged over 65 is for over 60 year olds due to data availability.

	1995			2005		
Pollutant: PM10	Low	Mid	Upper	Low	Mid	Upper
Cardiac morbidity >45	-40986	40489	59308	-25918	33325	52470
Cardiac morbidity	-45597	45044	65979	-35709	45915	72292

Table 41 - Hospital admissions due to car related air pollution based on Forbes ERC's – the low estimate is negative, which is a result of the range of uncertainty surrounding the coefficients. Also note that for 2005 the admissions data for those aged over 45 is for over 60 year olds due to data availability.

There is significant variation depending on the exposure-response coefficient chosen. There is some agreement between the APHEA results and the adjusted mean presented by Sahsuvaroglu and Jerret (for PM10). The Forbes (2009) coefficients however point to a much greater impact by air pollution, with a mid value of a 26% in morbidity per 10 µg / m^3 increase in PM_{10}. This could be a result of the study capturing long-term, chronic effects – when considering mortality, long term studies also captured a much greater impact than short term ones. However, the magnitude of impact on mortality, even when using long term studies, was much lower that the magnitude of impact presented in Forbes (2009), given that it was of 6% per 10 µg / m^3 increase in $PM_{2.5}$, which equates to roughly 12% per 10 µg / m^3 increase in PM_{10}. The range of uncertainty surrounding the Forbes

(2009) ERCs was also high – the low estimate was in fact negative (which would suggest health benefits from air pollution!). This is partly due to the fact that 2003 showed a much stronger association than either of the other years, which the authors were not able to explain. There is therefore some doubt on the results. It is worth mentioning both that the study employed the same pollution model used by COMEAP (2009) to determine pollutant levels, and that these estimates were lower than the ones used here, despite describing similar years (1994 and 2003). This could mean that our results are inflated. Finally, the Forbes ERCs only capture the impact on CV admissions, and not respiratory. Given there is significant evidence linking air pollution to respiratory disease, this would suggest the overall impact on morbidity is even greater. For the purposes of this study, the Forbes (2009) ERCs are nevertheless preferred, as the nature of the air quality data used (annual means) is better suited to capturing long-term impacts than short-term ones. It is however recommended that going forwards close attention be paid to cohort studies assessing the long-term impact of air pollution on morbidity, with the aim of providing a sounder scientific basis for morbidity ERCs.

5 .2. Impact of Road Accidents on Health

Accidents are a very visible impact of using private and light goods vehicles to provide mobility. Accidents affect both health (and thus the human/social store of value) and the economic store of value, due to the costs incurred by medical care and physical damage. Results are presented in both absolute terms and on a per passenger-km basis. This is perhaps less intuitive when considering accidents, which are linked to vehicle-km rather than passenger-km. However the goal is to assess the overall impact on the different stakeholders of the mobility consumed - hence using passenger-km is the correct metric.

When considering casualties, it is important to consider the impact on society as whole – as such, it is not sufficient to simply consider casualties among car users, but also casualties amongst other road users involved in accidents with private and light goods vehicles. Considering these casualties does not reflect which party was at fault in causing the collision – simply, it is a measure of casualties resulting from accidents involving private and light goods vehicles. These casualties would not have occurred if the private and lights goods vehicles did not exist, and hence are an impact of the fleet.

	1995				**2005**			
	Total:	Of which:			Total:	Of which:		
	Casualties	Killed	Serious Injury	Slight injury	Casualties	Killed	Serious Injury	Slight injury
Private & Light Goods Users	201227	1818	22749	176660	184350	1729	13475	169146
Per 100 million Passenger-km	32.08	0.29	3.63	28.16	25.38	0.24	1.85	23.29
Other Road Users	83792	1155	17224	65413	64656	948	11259	52449
Per 100 million Passenger-km	13.36	0.18	2.75	10.43	8.9	0.13	1.55	7.22
Total, all users	285019	2973	39973	242073	249006	2677	24734	221595
YOLL Due to Fatalities		118354				113694		

Table 42 - Casualties resulting from accidents involving private and light goods vehicles (DfT, 2006a and 2010) and resulting YOLL. 'Other road users' designate road users travelling in other modes (bicycle, pedestrian, goods vehicles, buses) involved in accidents with private and light goods vehicles.

Road Casualty Statistics (DfT 2006a, and DfT 2010, *Table 23*, *Accidents, vehicle user and pedestrian casualties: by combination of vehicles involved*) provide a breakdown of casualties among other road users involved in single vehicle or two vehicle accidents with private and light goods vehicles. They also include numbers for 3 or more vehicle accidents but do not detail the vehicle types involved. This means that a small fraction of casualties (other, non-pedestrian road users involved in accidents involving three or more vehicles and including at least one private and light goods vehicle) were not included in the results, presented in Table 42. Also included in Table 42 is an estimate of *Years of Life Lost* (YOLL) due to fatalities resulting from accidents. Population data and mortality data for 1995 and 2005 were again used in the IOMLIFET spreadsheets (Miller and Hurley, 2006) to provide estimated life expectancies for each age group. These life expectancies were multiplied by the number of road fatalities in each age group (table 34, *Casualties: by age, road user type and severity* in DfT, 2006a and DfT, 2010a) to give total YOLL. Per casualty, this equates to 43.3 years for car users, and 41 years for all road users in 2005. In 1995 these numbers are slightly smaller – 40.9 and 37.1 respectively, a change which is explained by longer life expectancies in 2005. Measuring YOLL allows for a better comparison to be made with impacts from air pollution, which was discussed in Section 5.1.2. However, it should be noted that despite the common unit, the comparison is not perfect – the impact from air pollution, though real and sizeable, affects a greater number of people by a small and not necessarily perceptible amount, as opposed to road casualties, which affect a smaller number of people by a much greater amount. Furthermore, YOLL do not fully capture the impact of accidents, as injuries, especially serious ones, significantly affect human welfare, both as a result of short term pain and suffering and long term disability. It is to partially address this issue that the World Health Organisation (WHO, 2012) makes use of Disability-Adjusted Life Years (DALY) to quantify impacts of health. DALYs are the sum of YOLL and Years Lost to Disability (calculated by multiplying, for each incident, the average duration of the health issue by an appropriate disability weight). This measure captures the impact of the burden of disease, and would allow a better comparison with morbidity due to air pollution. However, it requires more specific data on the nature, severity and duration of the health issues than were available for this study. Table 43 shows the change over the period, which is positive, both in terms of absolute numbers of casualties and in terms of casualties per passenger-km.

	Casualties	Of Which: Killed	Seriously Injured	Slightly Injured
Private & Light Goods Users	-8.4%	-4.9%	-40.8%	-4.2%
Per Passenger-km	-20.9%	-17.9%	-48.9%	-17.3%
Other Road Users	-22.8%	-17.9%	-34.6%	-19.8%
Per Passenger-km	-33.4%	-29.1%	-43.6%	-30.8%

Table 43 - Change in Casualties and Casualty Rates per Passenger-km, derived from Table 42.

5 .3. Impact of Noise on Health

Just as pollutant emissions cause a significant health impact, so do noise emissions. They are influenced by a myriad of factors – predominantly speed and volume of traffic flow, but also vehicle type, tyre type, level of maintenance, road surface and more. For an accurate assessment of the health effects, the level of exposure needs to be calculated, and combined with a measure of the corresponding health effect. Unfortunately, the availability of data is more restricted than for air pollution, although this has recently changed, with the EU Environmental Noise Directive (2002/49/EC) and subsequently the Environmental Noise (England) Regulations of 2006 requiring that noise maps be produced for major cities and roads detailing exposure to different noise sources. The UK National Noise Incidence Study 2000/2001 provides some insight into changes in noise levels, but does not differentiate between sources of noise. This means that it is of limited use for calculating the impact of private and light goods vehicles. Table 44 shows the results from the Noise Incidence Study. These show an improvement in day-time noise levels, but a worsening of night-time levels.

		Proportion of pop. exposed to noise exceeding guideline levels		
	WHO guideline	UK (2000/2001)	England & Wales (2000)	England & Wales (1990)
Day-time LAeq,16hr	55dB	54±3%	55±3%	60±3%
Night-time LAeq,8h	45dB	67±3%	68±3%	66±3%

Table 44 – Data on population exposure to noise from the UK Noise Incidence Study (2000), in terms of proportion of population exposed to noise levels exceeding WHO guideline levels (1990 and 2000).

Some studies, such as INFRAS/IWW (2004) and Maibach *et al.* (2008) focus on calculating the external costs of transport, and provide valuations of the costs associated with transport noise. These costs are based on a combination of the ‘Willingness To Pay’ to avoid noise disturbance, a valuation of the fatalities from cardiac infarctions caused by transport noise, and the medical costs associated with treating diseases caused by noise above 65 dBA. These costs are provided on a per vehicle-km basis for the year 2000. These can be used to estimate the impact of noise in monetary terms. The costs were applied to UK private and light goods vehicle-km in 1995 and 2005. Table 45 summarises the results. The costs per km were provided in terms of urban and interurban vehicle-km, so breakdown of UK vehicle-km by road type was used (available in DfT 2007a) to calculate the total costs for each vehicle type.

	Per 1000 Urban vehicle-km	Per 1000 Interurban Vehicle-km	1995	2005
Car	0.087	11.49	1691.1	1791.9
LGV	0.44	57.47	1071.8	1412.2
Total	-	-	2763.0	3204.1

Table 45 - marginal costs of noise, £million, including WTP to avoid disturbance, medical costs and valuation of fatalities

These are only rough values – a more accurate study would consider speed, volume of traffic and location, as well as time of day. It should also be noted that noise is logarithmic in nature and that halving or doubling the amount of traffic causes the noise level to change by 3 dB, regardless of the existing flow, which makes calculating noise costs based on the overall number of vehicle-km inherently inaccurate.

5 .4. Summary and Conclusions

In terms of health impacts, the picture is generally positive. The improvement in air quality has resulted in fewer deaths attributable to air pollution, as well as fewer YOLL. The impact on morbidity is somewhat less clear, as it varies with the ERCs that are chosen. The Forbes (2009) ERCs suggest a small increase in morbidity attributable to the car fleet, whilst others suggest a reduction. Despite the increased share of primary PM relative to national emissions, increasingly stringent emissions regulations appear to have driven a significant reduction in car fleet emissions. In terms of road safety, there is a marked improvement across all areas despite the large increase in passenger-km. Road safety remains the biggest cause of transport related mortality, causing approximately three times as many YOLL as emissions. Nevertheless, the impact of emissions is non-negligible, and should arguably be more widely recognised as a significant cost to car based transportation. Emissions of noise were also considered, although it was not possible to calculate their impact to the same degree of detail as for air emissions. It is likely that their impact has increased due to increases in annual vehicle mileage. This area warrants further examination.

Chapter 6

Economic Impacts of the UK car fleet

The existence of the car fleet, its operation, and the mobility provided impacts the economy in a variety of ways. The automotive industry is an important contributor to the economy. The government generates revenue from vehicle licensing and fuel duty. The mobility generated by the fleet enhances economic activity, although this effect is harder to quantify. Finally, the health impacts created by the fleet generate economic costs. All figures are inflation adjusted to 2005 pounds sterling to ensure a consistent basis for comparison over the period. This was done using the Retail Price Index (RPI). The conversion from 1995 to 2005 is approximately 1.288.

6 .1. Direct impacts of fleet operation

The manufacture, sale and operation of the UK car fleet require a large industry to support it, which directly impacts the economic store of value. The UK automotive industry saw overall growth over the study period, except in manufacturing, which underwent significant decay. The results, in terms of gross value added, are displayed in Table 46 and were adapted from the Annual Business Inquiry (ONS 2007a). Given that the data from ONS (2007a) relate to the entire motor industry, the values were corrected to better reflect the contribution of the private and light goods fleet. Different corrections were applied to different sectors:

- Manufacture and retail were adjusted according to the proportion of UK vehicle production manufacturing private and light goods vehicles (98%).

- Maintenance and sale of parts were adjusted according to the proportion of UK vehicles in the private and light goods class (89%).
- Fuel use was adjusted according to the proportion of UK fuel consumed by the private and light goods class (74%).

The strongest areas of growth were maintenance and sale of fuel. This is to be expected given the overall growth of the fleet – indeed, the growth in sales and maintenance of 25% is consistent with the 28% growth in fleet size. It should be noted that over the period considered, annual production of private and light goods vehicles increased from 1.731m to 1.768m (ONS, 2006), but that value added by manufacturing dropped, indicating that the sector became substantially less profitable. Additionally the number of new vehicles registered annually went up from 2.024m to 2.604m (DfT 2007a), which far outstrips the increase in production, meaning that a greater proportion of vehicles are being imported to the UK.

	1995	2005	Change
Manufacturing	11997	8906	-25.8%
Sales (vehicles, parts, fuel), and repair	16381	20563	25.5%
Total	28377	29469	3.8%

Table 46 - Approximate gross value added by the UK automotive industry, 1995 and 2005, £million.

Another economic impact of the car fleet is the tax revenue collected by the government through additional taxes such as Vehicle Excise Duty (VED) and the Hydrocarbon Oil Duty[8]. These taxes are extra to ordinary taxation such as VAT – as such their impact is not reflected by the gross value added by the industry (Table 47). The fuel duty figure is based on the fuel duty rates (HMRC, 2011) multiplied by fuel consumption (DfT 2007a), and the VED is taken from the DVLA accounts (DVLA, 2006) and the Appropriation Accounts

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[8] VED is often incorrectly referred to as ‘road tax’. Similarly, the Hydrocarbon Oil Duty is often referred to as ‘fuel tax’

1995-1996, a report to the House of Commons (HoC, 1997). Adjusted for inflation, it appears that the duty on petrol increased only slightly. This is due largely to a sharp increase in the proportion of diesel vehicles (from about 17% to 28%), which resulted in a slight decrease in the amount of petrol sold. Nevertheless, revenue still increased, which can be explained by the fuel duty escalator, brought in by the Conservative Government in 1993, which increased fuel duty by 3% ahead of inflation. This was eventually scrapped in 1999, partly as a result of protests from users. Revenues from diesel increased dramatically, boosted by the rise in popularity of predominantly diesel light goods vehicles (light goods vehicles in 2005 were 90% diesel). Overall, fuel duty revenue increased 24.5% despite an increase in fuel consumption of only 2.6%.

	1995	**2005**	**Change**
Petrol duty	12490	12548	0.5%
Diesel duty	2224	5779	159.8%
Total fuel duty	14714	18327	24.5%
Vehicle Excise Duty	4822	4588	-4.8%

Table 47 - Additional tax revenue raised due to private and light goods vehicles, £million.

Once adjusted for inflation, revenues from VED appear to have dropped somewhat. This is interesting, as during the time period the fleet size increased by 28.6%, which would be expected to increase revenues of VED. It is worth noting that in the intervening years VED switched from being a rate based on engine size (with only two bands available), to a more complex one based on CO_2 emission bands to incentivise the purchase of less polluting vehicles. This has to a certain extent worked – it was shown in Chapter 3 that although CO_2 emissions rose, they did not rise as much as vehicle-km, which means that CO_2 emissions per km dropped. However this has also resulted to reduced VED revenue. It is possible given the increase in fuel duty that this was a deliberate policy shift by the government, choosing to target the actual causes of pollution (fuel consumption and fuel efficiency) rather than engine size.

Finally, it is worth considering the impact that motor insurance has on the economy. Given that every driver has a legal requirement to be insured, this is directly attributable to the car fleet. Table 48 gives the total value of premiums sold and total outgoings of the motor insurance industry, taken from CEA[9] (2006) and CEA (1998). 1995 figures have been adjusted for inflation, and the share attributable to private and light goods vehicles is based on the proportion of UK vehicles in the private and light goods class. Although this is not a completely accurate method of attributing share (heavy goods vehicles and motorcycles will have different insurance premiums to cars) it is likely to be fairly close given that the overwhelming majority of vehicles on the roads are in the private and light goods class. The sector saw some growth although not as much as might be expected given the increase in fleet size over the period. It should be noted that although in 2005 outgoings exceeded income, the industry remained profitable due to income from investment. It was not possible however to find any numbers for profitability or value added, which would have allowed for a better comparison with the automotive industry, and provided a better measure of sector health.

	1995	**2005**	**Change**
Total motor premiums	10212.4	10421	2.0%
Total outgoings	9932.3	10610	6.8%
Share attributable to Private and Light Goods*	89.6%	88.8%	
Premiums	9147.2	9258.0	1.2%
Outgoings	8896.4	9425.9	5.9%
*based on number of vehicles registered			

Table 48 - Net Written Motor Insurance Premiums and Outgoings, £million.

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[9] 'Comité Europeen des Assurances' – this is the European insurance and reinsurance federation

6 .2. Economic Impact of Mobility Provided

Although quantifying the economic impact of the car-based mobility in monetary terms is difficult, some proxy indicators are generated to illustrate the economic importance of the mobility provided by the car fleet to the UK. These are displayed in the tables below. Table 49 shows the distance covered by purpose that can be attributed directly to economic impacts (adapted from NTS, 2006). NTS (2006) does not provide data for business mileage attributable to light goods vehicles. It was assumed that privately owned light goods vehicles cover the same proportion of business and commuting miles as privately owned cars, and that company owned light goods vehicles cover the same proportion of business and commuting miles as company cars. However a higher proportion of light goods vehicles are registered to companies, so a typical light goods vehicle has higher annual business mileage than a typical car. This was accounted for in the calculation. Table 50 shows the cost per km. The cost per km is calculated by dividing estimates for value of time, taken from DfT (2011b), by the average speed discussed in Chapter 4. The estimates for value of time were generated by DfT (2011b) for different modes of transport, and are split into values of working and non-working time. DfT (2011b) calculates the value of working time on the principle that time spent travelling during the working day is a cost to the employer's business, and the assumption that savings in travel time convert non-productive time to productive use. The actual value of this productive time is based on the average wage earned by users of the different modes. DfT (2011b) calculates the value of non-working time based on the fact that people implicitly place a value on their own time, and will trade a slower cheaper journey for a faster and more expensive one. However, exactly how much they are willing to pay depends of factors such as income, the value and urgency of the journey, and the comfort and attractiveness of the journey itself. The estimates in DfT (2011b) for this value of time account for these aspects, as otherwise transport investment would be biased, for instance towards projects benefiting wealthier individuals who are willing to pay more for faster travel. The value of non-working time does not differentiate by mode.

Using these values of time provides a means of recognising the economic benefit of faster travel – hence the lower the cost the better. Given the overall drop in speed over the time period, it is logical that the costs should have gone up slightly. The costs were not applied to the annual vehicle-km as this was deemed to be somewhat artificial, given the uncertainty surrounding the average speeds and the fact that it is not a direct cost, but better described as an opportunity cost – if the individual were not in transit, he or she would potentially be able to be productive or perform some other activity.

	1995		**2005**		**Change**	
	Total (billion vehicle km)	Per vehicle (km)	Total (billion vehicle km)	Per vehicle (km)	Total (%)	Per vehicle (%)
Business km/yr	72.5	3191	62.9	2151	-13.3%	-32.6%
Commuting km/yr	116.9	5145	145.4	4976	24.4%	-3.3%

Table 49 - Indicators of economic importance of mobility - distance travelled by purpose, adapted from NTS, 2006.

Business mileage noticeably reduced, suggesting that private and light goods vehicles are playing less of a role in supporting business. In terms of commuting, car mobility increased, although not on a per vehicle basis. This suggests that the popularity of private and light goods vehicles for commuting is also waning. These results are consistent with the results in Table 50 – if costs per km are increasing, it is logical that less use would be made of private and light goods vehicles for purposes related to economic gain.

	1995	**2005**	**Change**
Working time	0.48	0.50	9.1%
Commuting	0.11	0.12	4.2%
Other	0.10	0.10	0.0%

Table 50 - Cost per km based on value of time and average speed, £/km.

6 .3. Economic Impacts of Health Issues

Health impacts have an economic effect in terms of both the costs of treatment, and the costs associated with lost productivity.

6 .3.1. Accidents

Values expressing the economic impact of accidents were taken from DfT (2011b), which is a Department for Transport document outlining the methods used to appraise transport interventions. It includes means of evaluating the average cost of preventing an accident or casualty. The values are normalised to 2005 £ sterling, to allow for a clear comparison. The values contain both a measure of physical costs (such as healthcare) and measures of human costs, which encompass an estimate of the willingness-to-pay to avoid grief and suffering. These costs are presented for reference, but are excluded from the final total as human costs are deemed to affect the human/social store of capital rather than the economic one, and are as such accounted for in non-monetary terms (see Chapter 2). Costs can be appraised either on a per casualty or on a per accident basis. For the purposes of calculating economic aspects it is more appropriate to use data on a per accident basis, as this includes police costs, insurance costs and damage to property. However, given that the data for casualties were more detailed than those for accidents, a hybrid approach was used in this study. Costs dependent on the number and severity of casualty (i.e. medical, lost output, and human costs) are calculated on a per casualty basis. Costs dependent on the number and severity of accidents (i.e. police, insurance and damage costs) are calculated on a per accident basis. The final results are shown in Table 51. It is clear that although there was an improvement over the time period, due to the reduction in casualties, accidents still represent a significant impact, of the same order of magnitude as the tax revenue from Vehicle Excise Duty, and around 10% of the gross value added by the automotive industry.

	1995	**2005**	**Change**
Lost output	2,693	2,219	-17.6%
Medical and ambulance	664	472	-29.0%
Police cost	21	18	-13.0%
Insurance and admin	24	21	-14.8%
Damage to property	655	568	-13.3%
TOTAL	4,057	3,298	-18.7%
Human costs	*10,263*	*7,815*	*-23.9%*

Table 51 - Total value of prevention by element of cost: casualty costs (output, medical and human) calculated on per casualty basis, accident costs (police, insurance, damage) calculated on a per accident basis, £million.

6 .3.2. Air emissions – Health

Chapter 5 discusses the impacts of air emissions on human health. These impacts also have an economic component. Two types of impact were considered – morbidity and mortality. Evaluating the economic cost of mortality due to emissions is difficult – most approaches focus on the human cost aspect, and hence willingness-to-pay, rather than costs associated with lost productivity. This is understandable, given how air emissions affect health – the impact on an individual life is likely to be felt in months not years, and as such lost productivity is much harder to assess than it is for car accidents for instance. Therefore it was deemed more useful to focus on the morbidity impact, which was measured in terms of hospital admissions. These represent real costs – time spent in hospital costs money in terms of care and lost output. The values used were taken from Stieb *et al.* (2002) and are presented in Table 52. It should be noted that Stieb *et al.* were calculating costs for Canada, and as such may not be entirely appropriate to the UK. However, comparison with a study by the Department of Health (DoH, 1999) and a European study on external costs of transport (ExternE, 2005) showed that these values are sufficiently close to the UK situation.

	Cost of Treatment	Lost Productivity	Total Cost Estimate (including WTP)	95% CI
Respiratory Admission	1343	144	2015	[1631, 2399]
Cardiac Admission	1823	130	2495	[1919, 3070]

Table 52 - Cost of Treatment, Lost Productivity, and Total Cost (including WTP to avert pain and suffering) per hospital admission, from Stieb *et al.,* 2002, converted to 2005 £.

Table 53 presents the results of these values applied to the hospital admissions estimates from Chapter 5. Only the mean values were used – similarly only the mean cost was used from Table 52. Given the fact that both the hospital admissions estimates and cost estimates come with a large degree of uncertainty (expressed through the confidence intervals), the values in Table 53 are not particularly reliable – however, even when using the highest estimate from Forbes (which produced the largest admissions estimates) and the highest cost estimate, the result is £221.9 million, which is still an order of magnitude smaller than the costs attributable to accidents. This shows that from a purely economic perspective, the largest costs can be avoided by reducing road accidents, and that the direct impact of emissions on morbidity is not as important an issue.

	1995	2005	Change
Jerret & Sahsuvaroglu (PM10 only) ERC's	13.0	12.3	-5.5%
Forbes ERC's	87.9	89.6	1.9%
Aphea ERC's	6.0	5.6	-6.5%

Table 53 - Cost of treatment and lost productivity due to hospital admissions attributable to car emissions (based on mean admissions estimates, costs adjusted for inflation), £million.

6 .4. Summary and Conclusions

The overall picture for the sustainability of the car fleet from an economic perspective is a positive one. Most areas are moving towards a more sustainable situation, especially with respect to costs arising from health effects. In terms of economic impacts arising directly from the fleet itself, growth has not always been in line with growth in fleet size, notably in insurance, Vehicle Excise Duty and manufacturing. The latter two areas were the only two to worsen over the study period. However, revenue from fuel duty rose steeply despite only a small increase in fuel consumption, which more than compensated the slight drop in VED. It is therefore likely that this is due to a realignment in Government strategy. Manufacturing on the other hand showed signs of trouble. Production increased only slightly over the period, but more importantly value added dropped significantly. This means that not only is the sector showing limited growth, its profitability is decreasing.

The metrics chosen to describe the contribution car-based mobility makes to the economy suggest that private and light goods vehicles in 2005 are making a smaller direct contribution than they did in 1995. Overall business mileage decreased, as did per vehicle commuting distance. This could be partly explained by the fact that average speed dropped, reducing the benefit the car fleet provides, and increasing time-related costs of use. It nevertheless still performs an important role as an aid to commuting.

Chapter 7

Indicator Aggregation

As discussed in Chapter 1, aggregating a multitude of indicators into a single index provides significant benefits to a sustainability assessment by combining changes in sustainability into a single, easily understood number, whilst simultaneously raising a number of issues, linked principally to the processes of weighting and normalising the indicators. Additional issues include loss of information, as well as an implied 'substitutability' between different aspects of sustainability. However, as part of a holistic assessment, aggregation can be an important tool for summarising the results of an assessment, and drawing out the overall trends. It is in this capacity that it is undertaken here. No attempt is made to reduce the complex issue of sustainability to a single, all encompassing number; rather, the focus is on putting the different indicators into context, and understanding how they might relate to each other in terms of scale of impact.

7.1. Methodology

There are two aspects to aggregating the indicators generated. The first is how to normalise the various indicators, the second is how to weight them.

7.1.1. Normalising

There are a number of methods for normalising indicators depending on what the aggregation and the study are looking to achieve. Nardo *et.al.* (2008) highlight a number of them in their guide to composite indicator building. For the purposes of the study undertaken in this thesis, namely looking at the sustainability of the UK car fleet, the focus is on examining the changes in sustainability over the time period. Given that the aggregation is to be used as a tool to summarise these changes, rather than create some largely arbitrary 'index of sustainability', the approach used to normalise the indicators is to convert the change between the two years to a per cent basis. This is a simple, transparent and effective way of combining multiple disparate indicators. It expresses the direction the system is going with respect to sustainability. It does not provide an absolute measure of the sustainability of the system. As such a comparison to other systems (transport or otherwise) is only possible in terms of rate of change in sustainability. Equally, it anchors the analysis to the two years upon which the study is based – a subsequent analysis or update would have to refer back to the earlier studies. This would potentially create issues as later studies may wish to update the indicator set, depending upon how the system is changing, and what new data become available. Nevertheless, in the context of summarising the changes in sustainability across the study period, this method of normalising is effective.

7.1.2. Weighting

The weighting process is the most subjective, and hence controversial step in the construction of a composite indicator. A variety of methods are discussed in Nardo *et al.* (2008), although most are designed to aid in ranking a large number of countries across a set of indicators. One of the more appropriate for this study is the Analytic Hierarchy Process developed by Saaty (1990) which is used in several papers attempting to construct sustainability indices for industrial processes, such as Singh *et al.* (2007) and Kahn *et al.* (2004). Nardo *et al.* (2008) summarise the method thus:

> *"The core of AHP is an ordinal pairwise comparison of attributes. For a given objective, the comparisons are made between pairs of individual*

> *indicators, asking which of the two is the more important, and by how much. The preference is expressed on a semantic scale of 1 to 9. A preference of 1 indicates equality between two individual indicators, while a preference of 9 indicates that the individual indicator is 9 times more important than the other one. The results are represented in a comparison matrix [...], where Aii = 1 and Aij = 1 / Aji. [...] The relative weights of the individual indicators are calculated using an eigenvector. This method makes it possible to check the consistency of the comparison matrix through the calculation of the eigenvalues."*

Although the method provides a good way of weighting multiple criteria, it still relies on expert or stakeholder opinion to perform the weighting. Another option was considered, which relies on weighting the indicators according to an estimate of their impacts in financial terms. An example calculation which bases the financial estimate on the value of the 2005 indicator is given below. Indicator a would be calculated thus:

$$a(\%) = \pm \frac{a_{2005} - a_{1995}}{a_{1995}}$$

and its respective weight w_a:

$$weight, w_a = \frac{monetary\ value\ of\ indicator\ a\ in\ 2005}{total\ value\ of\ all\ indicators} = \frac{v_{a,2005}}{\sum v_{2005}}$$

where ± is dependent on the nature of the indicator (benefit or dis-benefit) and $v_{a,2005}$ is the value of indicator a in 2005, expressed in monetary terms. This ± is included to ensure that a positive change in a describes movement towards sustainability, and conversely that a negative change describes movement away from sustainability. Thus, in the event the indicator describes a dis-benefit, the – is applied. This ensures, for example, that a negative change in an indicator describing a dis-benefit results in a positive change for sustainability.

This method is subtly different to a simple cost-benefit analysis in that it is not an account, and does not create a bottom line, but instead uses financial valuation a means of estimating the scale of each impact.

7 .1.3. *Procedure*

In order to apply the method described above, a monetary estimate must be made of all the indicators considered in the study. The following sections will therefore generate monetary estimates of the impacts described by the indicators calculated in chapters 3, 4, 5 and 6. These estimates will be used as weights for aggregating the indicators in the manner outlined in the equations above, resulting in an overall percentage change in the car fleet's sustainability between 1995 and 2005.

7 .2. Environmental Indicators

The valuation of environmental impacts is a controversial issue, but has been the subject of a wealth of studies. Famously, the Stern Report (Stern, 2007) attempted to put a financial cost on climate change. A difficulty with the current study is that most past studies examining the external costs of air emissions focus on the impacts to human health. Given that in this study these impacts are considered separately from the environmental issue, it is challenging to find estimates of costs that do not cause double counting. This is true of the impact of particles and photochemical oxidation. It is also true to some extent of the impact of climate change, but this has not been considered in the Human/Social health chapter (Chapter 5) so it is appropriate to consider it fully here.

A number of different studies were considered to perform the valuation of the impacts discussed in Chapter 3. For particles (PM_{10}), the value of a tonne of emissions was taken from Hamilton and Atkinson (1995), who provide figures for the impact of particles on health and on materials. Only the value for materials (~ £280/tonne) was used. For photochemical oxidation, the values were calculated from the CAFÉ project (AEA Technology, 2005), which calculated the impact of various ozone producing pollutants on crops. Based on the pollutants included in the photochemical oxidation indicator, a value of ~£37/tonne was calculated. Values of £1760 for acidification and £3087 for eutrophication were found in Leduc *et al.* (2008)[10]. The impact of waste to landfill was calculated on the basis of two studies. The first, Powell and Brisson (1994) estimated the amount of CO_2 and methane produced per tonne of landfill waste. This, multiplied by the cost of CO2 emissions, provides an estimate of the emissions impact of landfill. To this was added the cost of landfill disposal in the UK, taken from GHK (2006), as this is an additional economic cost not present for other emissions. The final value was £60 per tonne. The cost of energy resource depletion due to primary energy use is difficult to quantify, as it depends on availability of reserves, rate of production, and future energy demand, all of which are subject to uncertainty. The approach used here was to use the present energy price, using DUKES (DECC, 2006). 'Primary supply' in tonnes of oil equivalent was divided by the 'Basic value of inland consumption' (this is the cost of energy supply excluding tax and distributions costs and margins) to produce an average cost per unit of energy. Although this does not address the issue of future resource scarcity, it has the merit of being based on real world data on the value of energy in 2005. The cost of global warming is subject to a lot of debate. Leduc *et al.* (2008) review several studies on the costs of CO_2, and settle on a central value of 50€/tonne (£33). This is lower than the cost of damage determined by Stern (2007), which was $85 in 2000 prices. This works out at £53/tonne CO_2. Barker (2008) notes that many 'conventional' economists criticised the approach used by Stern, namely his choice of discount rates. Barker nevertheless supports the Stern values, centring his argument on the uncertainty surrounding the response of the

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[10] Many of the values were expressed in different currencies, and were converted to £Sterling using the appropriate historical exchange rates. This explains the lack of round numbers.

global economy to a potentially non-marginal change such as climate change, given it is a complex, non-linear dynamic system. He also argues that the issue of intergenerational equity, central to the climate change issue, is an ethical one and decisions should therefore not be informed by economics in isolation. For the purposes of this study, estimates using both values were calculated. The results, summing the impacts of vehicle life cycle and fuel use, are presented in Table 54.

	Nature of Indicator	**Change Over Period**	**Monetary Value, 2005**
Global Warming *(Leduc et al. 2008)*	Dis-benefit	10.7%	4000.0
Global Warming *(Stern, 2007)*			6431.1
Acidification	Dis-benefit	-39.3%	835.6
Eutrophication	Dis-benefit	-48.1%	178.4
Particles PM_{10}	Dis-benefit	3.9%	16.2
Photochemical Oxidation	Dis-benefit	-54.4%	12.3
Primary Energy Use	Dis-benefit	9.3%	9753.9
Waste to landfill	Dis-benefit	-23.5%	29.3

Table 54 - Change in economic indicators over study period, with monetary value, £million.

It can be seen from the results that by far the largest impacts in monetary terms are attributed to *global warming* and *primary energy use*, which are both an order of magnitude larger than the next largest impact, and two orders of magnitude larger than the smallest impacts. This is to understandable given the large volume of CO_2 emissions and quantity of energy used. Another contributing factor is the reliability of the values used. Global warming is an issue with a very high level of awareness, and has been the focus of a large number of studies. This means that the price of CO_2 used is likely to be more accurate than for some other impacts. Primary energy is a traded commodity – the values used were set by the market, and are thus reliable. In the case of the three smallest impacts (from *waste to landfill*, *photochemical oxidation* and *particles*) the story is somewhat different. The impact of particles was purely based on damage caused to materials by particles in the atmosphere. Even though the health impact of particles is considered elsewhere, it seems likely that the impact of particles on the environment is not limited to material damage. The same point goes for the impact of ozone formation. Only the impact on crops is considered by the monetary valuation, when other environmental impacts (impacts to forests and other vegetation for instance) almost certainly occur. Finally, in the valuation of waste, no value is placed on the land use aspect of landfill. It is therefore likely that all three of these categories are undervalued to a greater or lesser extent.

7 .3. Human/Social Indicators

7 .3.1. Weighting Mobility

Putting a monetary value on mobility, and the associated service quality characteristics, requires an overall estimate of the value attached to the service. The approach used here is to consider the value of the mobility as being equal to what the population is willing to spend on it. This will provide an estimate of the total value attached by its users to the combination of distance, service quality, and access (equality of opportunity) that the private and light goods fleet provides. It will allow the sum of those human/social impacts to be weighted relative to others, such as health, environmental and economic impacts. A further weighting process will be required in order to weight the indicators of mobility, service quality and equality of opportunity relative to each other. This will not however be done using monetary valuation.

ONS statistics on household expenditure (ONS 2007b) were used to account for all private expenditure on private and light goods transport. HMRC mileage allowances[11] were used to estimate the business expenditure on private and light goods transport, based on the number of business vehicle-km. It should be noted that the mileage allowance reduces after the first 10000 miles. However DfT transport statistics (DfT 2007a) show that the average annual business mileage for company cars, the portion of the fleet with the highest number of business miles, was approximately 7500 miles in 2006, suggesting that few vehicles exceed this limit. It should also be noted that these allowances are applicable to business miles covered in privately owned vehicles, and not company cars. However, they serve as a useful estimate of the cost-per-mile to businesses. Table 55 shows the individual components of transport related household expenditure in 2005, adapted from ONS 2007b.

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[11] The cost per mile that businesses are allowed to claim as business expenses, HMRC 2012

Some categories were not specific to cars and vans, and had to be adjusted to remove the contribution from motorcycles. Motorcycles account for 3.5% of insurance and ‘other costs’ (which include RAC costs, driving lessons, parking and sundry other expenses) and 0.8% of petrol costs (0% of diesel). These percentages were therefore subtracted from the ONS data, yielding the corrected values displayed in Table 55. The insurance and ‘other costs’ corrections were based on vehicle numbers and the fuel correction on fuel usage data (DfT 2007a).

From section 6 .2, we know that the distance travelled for business in 2005 is approximately 63 billion vehicle-km. When multiplied by the HMRC allowance of 40 pence per mile (~25p per km), the total expenditure for business mileage is £15.6 billion. Therefore the sum of the household and business expenditure on private and light goods based transport is £83.9 billion. This is an estimate of the total value attached to mobility, and includes the value attached to distance, service quality and equality of opportunity provided by the fleet.

Category of transport-related household expenditure		**Total weekly expenditure**
Purchase of cars and vans	New	237
	2nd hand	347
Operation	Spares	43
	Fuel*	430
	Repairs	17
	Other costs*	56
Insurance*		177
Total weekly household expenditure		1308
Total annual household expenditure		68237
*these values were adjusted to allow for motorcycles		

Table 55 - Total annual household expenditure on private and light goods vehicles, adapted from ONS (2007b), £ million. These data as reported by ONS specifically exclude business travel expenditure to avoid double counting.

7 .3.2. Weighting the Components of Mobility

The estimate of the total value attached to car-based mobility calculated above does not allow the individual components of mobility (distance, service quality, equality of modal opportunity) to be weighted relative to each other. This would be difficult to achieve using monetary valuation, as it would require pricing quality characteristics versus mobility, as well as pricing each component of service quality individually. The literature, whilst helping to identify the issues of importance to users, does not provide any guidance on actual weights. The components were therefore weighted in a hierarchical fashion according to the author's opinion, making use of the literature to support the weighting process, as well as the AHP where necessary. Based as they are on the decisions of a single expert, the weights produced are highly subjective, and would be better supported by wider stakeholder consultation or survey. The reason for this is twofold. Firstly, a statistically constructed survey would produce weights that better represent society's preferences. Secondly, it would address the issue of researcher positionality biasing the research by removing the researcher from the weighting process. This could be achieved by even a limited survey. It is suggested that addressing these issues would be a priority for future work.

The hierarchy used is shown in Figure 12; this illustrates how the stages of the aggregation are broken down into tiers. The indicators in tier 1 are the ones discussed in Chapter 4, with the exception of employment, which will be included in the section on economic impacts. For the first stage (tier 1 to tier 2), the weights were assigned without use of the AHP. Journey time was split into 60% speed and 40% proximity to origin/destination, partly due to data quality considerations (the data on speed being more reliable). The components of comfort were all weighted equally – although a review of market research could reveal which aspects of comfort are most valued by customers, the comfort element as whole was subsequently weighted low enough that individual aspects would not have a significant influence on the overall result regardless of their respective weights. Finally, the elements of 'equality of opportunity' were weighted as 50%, 35% and 15% for affordability, access to licenses and blue badges respectively.

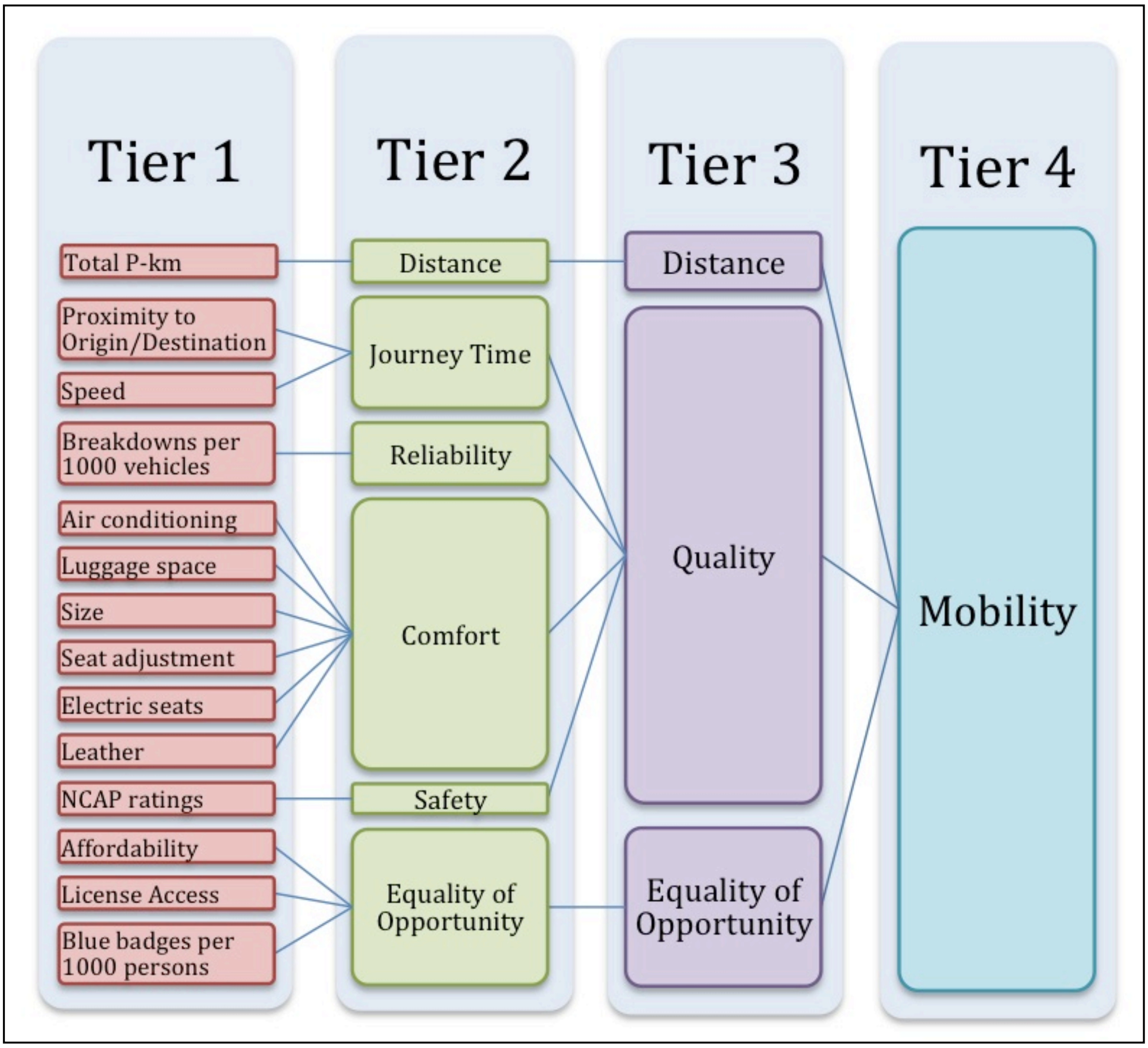

Figure 12 - Hierarchy used for aggregating mobility-related human/social indicators

For the second stage (tier 2 to 3) the Analytic Hierarchy Process was used, as the relative importance of individual aspects of service quality was harder to discern. This yielded the weights displayed in Table 56. The dominance of safety over other aspects may seem counter-intuitive – when considering the advantages provided by the car, the service aspects that most easily come to mind are speed, convenience and reliability

Service Quality Aspect	Weight
Journey Time	12%
Reliability	26%
Comfort	6%
Safety	57%

Table 56 - Weights obtained for elements of service quality using the AHP.

Nevertheless, when going through the process of making pairwise comparisons, safety came first, as evidenced by the significant weight (57%) attached to it. This may be due to the implicit assumption that the safety of the car is 'sufficient' being challenged by the process of going through the AHP and comparing the importance of safety with other service characteristics. The fact that the service characteristics being weighted (safety, reliability, speed, comfort) are not specific to cars, but are applicable to transport in general, may further remove any bias in the perception of car safety. It should be noted that a multitude of factors have an influence on risk perception in transport, such as perceived control, actual risk, severity of outcome (in the event of an accident), age, education, gender, risk sensitivity, and 'newness' of the technology (Rundmo *et al.*, 2011). The perception of control when using private transportation tends to reduce the perception of risk. Nevertheless, it bears repeating that this weighting is entirely subjective, and is quite likely to change if a wider pool of experts and stakeholders were surveyed - the dominance of safety could be a result of personal bias on the part of the author. It is however quite likely that the overall order of preference would remain the same, if not the values of the weights.

Weights assigned to Distance, Quality and Equality of Opportunity	Change in Human/Social Impact from 1995 to 2005
40%, 40%, 20%	21.1%
20%, 40%, 40%	20.2%
Equal	19.5%
45%, 40%, 15%	21.3%

Table 57 - Resulting changes in Human/Social impact from 1995 to 2005, depending on the emphasis placed upon the three components mobility (distance, quality, and equality of opportunity).

For the final stage (tier 3 to 4) a number of different weights were tried, each emphasising different aspects of transport, partly as means of testing the sensitivity, and partly to explore how different perspectives on transport might alter the perception of the contribution of the car fleet to human/ social capital. As can be seen from Table 57, the different weights only alter the result by a percentage point (approximately 5%) either way. For the purpose of this study, we suggest weights of 45%, 40%, 15% for distance, quality and equality respectively. This assignment of weights recognises that the provision of mobility is the foremost purpose of the car fleet, and hence the quantity, and to a slightly lesser extent the quality, of the mobility provided should be the main focus. The final weights assigned to each indicator, assuming this weighting, are displayed in Table 58 for reference purposes. As a reminder, the weight assigned to mobility as a whole (tier 4) is the monetary estimate resulting from the calculation performed in section 7 .3.1, £83.9 billion.

Indicators	w_i	**Tier 2**	w_{t2}	**Tier 3**	w_{t3}
Total P-km	45.00%	Distance	45.00%	Distance	45.00%
Proximity to Origin/Destination	1.88%	Journey Time	4.70%	Quality	40.00%
Average Speed	2.82%				
Breakdowns per 1000 vehicles	10.49%	Reliability	10.49%		
Air Conditioning	0.37%	Comfort	2.21%		
Luggage Space	0.37%				
Size	0.37%				
Seat Adjustment	0.37%				
Electric Seats	0.37%				
Leather interior	0.37%				
NCAP Ratings	22.60%	Safety	22.60%		
Affordability	7.50%	Equality of Opportunity	15.00%	Equality of Opportunity	15.00%
License Access	5.25%				
Blue Badges per 1000 persons	2.25%				

Table 58 - Weights assigned to individual indicators, assuming a [45%, 40%, 15%] Distance, Quality, Equality split.

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7 .4. Human/ Social Health Impacts

There are several different health impacts to consider:

- Morbidity and loss of life associated with air pollution,
- Injury and loss of life occurring as a result of collisions,
- Health impacts due to noise.

So far the indicators discussed in financial terms (with the exception of noise – see section 5 .3) have only considered the physical costs associated with the health impacts, such as medical costs or physical damage. However, given that an attempt is being made to weight these indicators relative to other indicators (such as those describing mobility-related impacts) using monetary valuation, the entirety of the impact must be considered. In order to do so, an estimate must be made of the value attached to loss of life, as well as to suffering or inconvenience caused due to illness or injury, more easily described as 'human costs'. Such estimates are based on the concept of willingness-to-pay (WTP) - attempting to evaluate what people are willing to pay to avoid certain impacts, or reduce certain risks. These valuations are controversial for moral and ethical reasons, and concerns over the validity of the results. These can vary significantly based on the methods used to perform the valuation.

Monetary valuation methods broadly fall into two categories: direct methods and indirect revealed preference methods. Direct methods make use of market prices and replacement costs. Since human costs do not relate to physical impacts affecting market goods, or to losses that can be replaced (or restored) with market goods, indirect revealed preference methods must be used. One commonly used method for air pollution in particular is the contingent valuation method, described in ExternE (2005) as:

> *“ a survey method in which respondents are asked to state their preferences in hypothetical or contingent markets, allowing analysts to estimate demands for goods or services that are not traded in markets.”*

ExternE (2005) and Desaigues *et al*. (2007) discuss some of the issues with contingent valuation methods, specifically when applied to mortality risks. These include:

- The fact that the sensitivity to changes in risk is typically poor, i.e. willingness to pay does not increase proportionally to an increase in risk. This suggests that respondents struggle to fully understand the meaning of small changes in probabilities,
- The fact that payments made are purely hypothetical, and thus largely exempt from real-world budget constraints,
- The fact that the scenario proposed by the survey is not always plausible to respondents or fully understood.

Other issues include variations in WTP due to age, the distance into the future at which impacts occur and context. ExternE (2005) notes that some studies have found WTP to increase with age, whilst WTP tends to decline for chronic impacts likely to have an effect in the future. Context also affects WTP – the value of a prevented fatality in the context of rail accidents for instance is higher than that in the context of road accidents, most likely due to the perceived degree of control. Finally, small changes in the methodology used can have a big impact - Desaigues *et al.* (2007) note that if the survey uses closed bids (i.e. if respondents are offered a choice of values for changes in mortality risk) there is a significant anchoring bias, meaning that the value of the first bid has a large influence over the final WTP.

7 .4.1. Air Emissions - Mortality

Mortality can be measured either in terms of *statistical lives lost*, or *years of life lost* (YOLL); monetary valuations have been estimated for both. Historically more studies have focused on the value of a statistical life (VSL) but it is increasingly accepted for impacts linked to air pollution in particular that the value of a life year (VOLY[12]) is more appropriate. This is because of the way air pollution affects health. It increases the risk of mortality across all ages, leading to lives being shortened a little, rather that causing acute mortality in the manner of a fatal collision. ExternE (2005) suggests using a VOLY of €50,000 (~£33,750) for chronic effects, and €75,000 (~£50,650) for acute effects. From Section 5 .1.2, ~51,000 years of life were lost due to air pollution in 2005, giving a value of £1.72 billion. This value can be used to weight the indicator in a similar manner to the Human/Social indicators related to mobility. This indicator is the change in 'years of life lost due to air pollution' between 1995 and 2005 (Table 59).

	Nature of Indicator	**Change Over Period**	**Monetary Value, 2005**
YOLL due to car-based air pollution	Dis-benefit	-18.4%	1724

Table 59 – Change in YOLL, %, with corresponding monetary weight, £million.

7 .4.2. Air Emissions – Morbidity

Air emissions affect health, causing an increase in hospital admissions. This creates both physical costs (i.e. loss of productivity and medical costs), and impacts most easily described as human costs, related to pain, suffering and inconvenience. Whereas these two aspects were previously treated in separate sections, the fact that an overall monetary estimate is being made means that it is more logical to deal with the whole impact at once. Using the data on hospital admissions due to air pollution calculated in Chapter 5 (section 5 .1.3), and the estimates of monetary costs associated with these admissions introduced in

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[12] The Value of Life Year, or VOLY, is essentially the monetary valuation of a Year of Life Lost, or YOLL.

Chapter 6 (section 6 .3.2), we can calculate a value for the cost of morbidity, which can be used as a weight. In Chapter 6 only the physical costs were considered, but now the estimate of WTP to avoid illness calculated by Stieb *et al*. (2002) is also included. Due to a significant variation in the values describing the morbidity impact of air pollution, depending on type of Exposure-Response function used, three different estimates were considered. The indicators, with their corresponding weights, are displayed in Table 60.

Source of ERC	**Nature of Indicator**	**Change in hospital admissions**	**Monetary Value, 2005**
Jerret & Sahsuvaroglu (PM10 only) ERC's	Dis-benefit	-5.5%	16.6
Forbes ERC's	Dis-benefit	1.9%	114.5
Aphea ERC's	Dis-benefit	-5.9%	7.4

Table 60 - Change in hospital admission due to air pollution, %, with corresponding monetary weights, £million.

7 .4.3. Casualties Due to Collisions

Road accidents, in a similar manner to morbidity, create both physical impacts (related to medical costs, damage, lost output, insurance and police costs) and human costs (related to grief, pain and suffering). However, unlike morbidity, some of these costs are dependent on the number of collisions, whereas others are dependent on the number and severity of casualties. Therefore the impact of road accidents will be considered across two sections. The costs that depend on the number of casualties (human costs, medical costs and lost output) will be considered here, whereas those that depend on the number of accidents (police costs, damage to property and insurance costs) will be considered in the following section. Table 61 displays the change in casualty numbers (from section 5 .2) and a monetary estimate of human costs, medical costs and lost output, based on the value of a prevented casualty taken from DfT (2011a).

Injury Severity	Users affected	Nature of Indicator	Change Over Period	Monetary Value, 2005
Slight	Private and light goods users	Dis-benefit	-4.25%	2157
	Other road users	Dis-benefit	-19.82%	1802
Serious	Private and light goods users	Dis-benefit	-40.77%	2463
	Other road users	Dis-benefit	-34.63%	1350
Fatal	Private and light goods users	Dis-benefit	-4.90%	2087
	Other road users	Dis-benefit	-17.92%	647
Total	All	Dis-benefit		10506

Table 61 - Change in casualties, %, and monetary estimate of human costs, medical and ambulance costs and lost output due to casualties from road accidents involving private and light goods vehicles, £million.

DfT (2011a) provides guidance for appraising transport interventions. As such, the values provided a primarily designed for use in appraising road schemes. For mortality, DfT (2011a) considers the value of a prevented fatality, which is in effect the same as the VSL. It does not therefore work on the basis of YOLL. In Chapter 5, an estimate of YOLL due to road accident fatalities was calculated, which worked out at approximately 41 years of life lost per road fatality. If we take the value of a life year (VOLY) used in the section on mortality due to air emissions (section 7 .4.1), a comparison can be made between the two methods, shown in Table 62. It can be seen that the DfT value is significantly lower than the estimate calculated using the VOLY for acute effects (which fatalities from road accidents are), especially considering that the DfT value includes medical and ambulance costs and lost output. The ExternE VOLY is based on willingness to pay to reduce mortality risk, which does not (explicitly) address those factors. This discrepancy between the two values suggests that mortality from road accidents could arguably be weighted higher. However, it should be noted that both sources are based on different studies, using different methodologies[13], in different contexts (air pollution and road safety), all of which can affect the results significantly. In light of this, the estimates are quite consistent with one another.

Source	VPF or VSL
DfT 2011a (includes Human costs, medical costs and lost output)	1.42
Using ExternE VOLY – Chronic mortality	1.38
Using ExternE VOLY – Acute mortality	2.08

Table 62 – Value of a statistical life as taken from DfT (2011a) and as calculated using ExternE (2005) VOLY, assuming 41 years of life lost per fatality, £million.

7 .5. Economic Indicators

Several economic indicators were discussed in Chapter 6. Since these were mostly expressed in monetary terms, the process of weighting them was considerably more straightforward. Some of the indicators from Chapter 6 were addressed in earlier sections, such as the impact of morbidity (section 7 .4.2). The direct contribution of the fleet to the economy in terms of business mileage is difficult to measure in monetary terms, other than in terms of its cost, which was accounted for in the monetary estimate of the value of mobility. Thus this was omitted from the analysis. The insurance industry impact was also omitted due to the fact the numbers calculated (premiums collected and total outgoings) do not express profit or value added as much as turnover. Using these numbers would proportionally weight insurance more heavily given that turnover is always greater than value added. Employment has been included here, as it was omitted from section 7 .3. The indicator for employment is number of jobs, and the value assigned as a weight is the total cost of employment (gross salaries and wages, and social security costs) in 2005. Finally, those accident-related costs that depend upon the number of accidents rather than the number of casualties are also included here. The indicators are all shown in Table 63.

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[13] DfT uses stated preference, whereas ExternE uses contingent valuation.

	Nature of Indicator	**Change Over Period**	**Monetary Value, 2005**
Automotive Industry – Value Added			
Manufacturing	Benefit	-25.8%	8906
Sales (vehicles, parts, fuel), and Repair	Benefit	25.5%	20563
Total	Benefit	3.8%	29469
Tax Revenue Collected			
Total Fuel duty	Benefit	24.55%	18327
Vehicle Excise Duty	Benefit	-4.85%	4588
Automotive Industry - Employment			
Manufacturing	Benefit	-27.3%	6482
Sales (vehicles, parts, fuel) and Repair	Benefit	0.9%	10143
Total	Benefit	-8.3%	16625
Accidents involving private and light goods vehicles			
Police costs	Dis-benefit	-13.0%	18
Insurance and admin. costs	Dis-benefit	-14.8%	21
Damage to property	Dis-benefit	-13.3%	568

Table 63 - Change in economic indicators over study period, with monetary value, £million.

7 .6. Discussion and Final Aggregation

By aggregating the indicators presented in sections 7 .2, 7 .3, 7 .4 and 7 .5 using the monetary estimates as weighting factors, we arrive at a change in sustainability between 1995 and 2005 of +12.7 %, suggesting that the overall sustainability of the fleet has increased. The aggregation (using the Stern value for CO2 and the Forbes ERCs) is shown in Table 64.

	Value, £ million	**Change over Period**	**Weight**	**Weighted Change**
Environmental	17255	-6.8%	9.4%	-0.6%
Human/Social Mobility	83900	21.3%	45.8%	9.8%
Human/Social Health	12345	19.9%	6.7%	1.3%
Economic	69616	5.9%	38.0%	2.2%
Total	183118			12.7%

Table 64 - Final aggregation: change in sustainability in the UK car fleet, 1995 to 2005.

Before delving into the specifics of what this result means for sustainable mobility (see end of section), one should consider the process by which the number was generated. Initially a set of indicators describing the impact of the car fleet on sustainability was generated using the PAM. This method transparently links indicators to impacts that create sustainability issues affecting various stakeholders in the system. This indicator set was then populated in a study of the UK car fleet in 1995 and 2005. Finally, these indicators were normalised to a percentage change across the time period, and aggregated using weights assigned according to monetary estimates of the impacts described. The process of aggregation necessarily involves a loss of information and richness in the analysis, and is best used in tandem with the detailed study in chapters 3, 4, 5 and 6. It is useful as a means of helping to summarise the changes, but not as a standalone result. Particular attention ought to be paid to the weighting method. The use of monetary valuations as weights yields different results from a conventional cost-benefit study, in that sustainability is not equated with profitability and there is no positive or negative bottom line. Nevertheless the method shares many of the problems of cost-benefit or external cost analysis, namely with respect to the validity of the values attributed to the impacts. This is especially true when considering morally and ethically difficult to value impacts such as environmental damage or the loss of a human life. The main body of the study explicitly chose to avoid such valuations, and expressed these impacts in their own units. For instance, human mortality was expressed in terms of years of life lost (YOLL). The value of a YOLL was treated independently from monetary considerations. There are several strong arguments for so doing. The first is the accuracy and validity of monetary valuations. Values for human life and health (section 7 .4) were determined through a variety of survey methods using the concept of willingness-to-pay. These methods are not particularly reliable, as the results produced can fluctuate significantly depending on the methods used. In general it appears that people struggle with performing such valuations, and are not particularly consistent in their responses. Equally, for valuations on life in particular, an ethical question is raised. Can a value ever be placed on human life? The controversy surrounding the Ford Pinto case (Dowie, 1977) is a telling example of the dangers of such valuations. In terms of valuing environmental damage, Rees and Wackernagel (1999) highlight six problems with the pricing of natural capital in their critique of the use of monetary analysis for sustainability. These are:

- *Biophysical scarcity* – physical scarcity is often poorly reflected in consumer prices. The issue is that prices are often set more by market availability than actual resource abundance. Other factors such as availability of alternatives and demand also affect market prices.
- *Discounting* – this biases monetary analysis against the future. Natural resources do not inflate or deflate indefinitely over time in the way money does.

- *Markets overriding local factors* – market fluctuations will affect the value of natural assets, even if these are unchanged.
- *Substitutability* – manufactured capital can usually not be substituted for natural capital, despite the fact that they are priced in the same way. Many natural resources are essential to life, and can not be replaced by human-made goods.
- *Monetary limitlessness* – there are no apparent limits on the growth of money. This is at odds with the fact that there are physical limits to material growth.
- *Critical natural capital stocks* – market values assume perfect knowledge and informed decisions on the part of producers and consumers. Furthermore, the nature of complex natural systems is such that when critical thresholds are reached, large, irreversible changes can occur. Prices do not reflect whether a resource is critical to the integrity of the ecosystem. Equally, natural goods and services for which there is no market do not have prices – even if they are critical.

These criticisms are valid, and support the decision not to use monetary valuation in the main body of the study. However, given that weighting demands a relative valuation of different impacts to be made, resorting to a commonly used and understood unit of value such as money is defensible, an argument echoed by Costanza *et al.* (1997) in their widely cited work on the valuation of ecosystem services:

> "*Some argue that valuation of ecosystems is either impossible or unwise, that we cannot place a value on such 'intangibles' as human life, environmental aesthetics, or long-term ecological benefits. But, in fact, we do so every day. When we set construction standards for highways, bridges and the like, we value human life (acknowledged or not) because spending more money on construction would save lives*"

This highlights the fact that monetary values are commonly, and unavoidably, placed on so-called ‘intangibles’, whether the process is made explicit or not. It is a strong argument in favour of using monetary valuation to aid in weighting different indicators: since valuation is unavoidable, a pragmatic solution is required. Monetary valuation methods offer an insight into how to value disparate impacts relative to each other. The emphasis on using the monetary valuation purely as a weight somewhat distances this aggregation from conventional cost-benefit analysis and the dangers associated with it, since ultimately what is being measured is whether the system is moving away from or towards sustainability.

The principal concern with the methodology used is that an incorrect or incomplete valuation leads to certain impacts being under- or over-represented in the aggregation. This issue becomes more prevalent in the event of a consistent degradation of an aspect of natural capital. The monetary values calculated for environmental and health issues are marginal costs, meaning that they reflect the cost of a small increase in a particular indicator. As such, they are only valid for small, incremental, changes. However, when critical natural capital stocks are being degraded to the point of depletion, the value of the natural resource effectively becomes infinite, rendering the marginal costs applied to it invalid. The impact would then effectively become ‘under-weighted’ in the aggregation. Another practical issue with the method is that the weights for different indicators are effectively subject to market fluctuations, and can change over time. Nevertheless it has provided a useful means of summarising the study findings, and displaying them in an easily understandable form.

The results presented in Table 64 show that the sustainability of the car fleet has increased by ~13%, corresponding to an annual increase of ~1.2%. This is an improvement, albeit a modest one. It is difficult to gauge whether this represents a slow or a fast rate of improvement. It is possible to look to some other indexes and indicators as a means of comparison, but given it is calculated using a unique transport specific indicator set and aggregation method, such a comparison is not particularly meaningful. One would have to perform several methodologically and philosophically similar studies on other systems to be able to make direct comparisons; otherwise one is at risk of comparing methods rather than systems. The Regional Index of Sustainable Economic Welfare (R-ISEW - Jackson *et al.*, 2008) measures a mix of economic, environmental and social factors in monetary terms. In the UK, the R-ISEW per capita increased by ~41% between 1995 and 2005. The Human Development Index increased by only ~5% (UNDP, 2012) although it should be noted that the UK was considered to have "very high human development" throughout the time period, suggesting the potential for improvement was more limited. On a purely economic level, GDP rose by ~97%, and GDP per capita rose by ~ 90% (World bank, 2012). In conclusion, the numbers indicate that there is a steady incremental trend towards a more sustainable car fleet, but the fact that certain issues such as congestion and CO_2 emissions are getting worse despite widespread awareness of the problems they cause indicates that there should be some concern over its long-term sustainability.

Methodologically, the aggregation appears to be biased towards 'man-made' capital, namely mobility and economic aspects, with mobility receiving a weight of 46% and economic impacts a weight of 38%It should be noted that these weights are not a matter of *choice*. The weights assigned were calculated reflect the magnitude of the impact under consideration (albeit measured in economic terms). If, for instance, the environmental impact were larger, it would have a greater weight. The main issue with these weights is whether the economic valuations used to measure the magnitude of the impact provide an accurate reflection of the impact. On this count, the bias could be (at least partly) explained by the fact that the valuations for the economic impacts and mobility impacts use real data on revenues, expenditure and value added and are thus more reliable than the estimates made to describe environmental and health impacts, which involve significant uncertainty (as discussed earlier in this section). The weights in Table 64 therefore raise the question of whether environmental and health impacts are currently undervalued, or whether their magnitude really is much smaller than that of the economic and mobility impacts.

It is useful to consider how different weights might affect the result. Were each of the four main aspects to be weighted equally, the change in sustainability over the period would only be +10.5%. Alternatively, were the two Human/Social sections combined into a single indicator, equal weighting between the Environmental, Human/Social and Economic domains would result in a change of +6.8%. It can therefore safely be concluded that the sustainability of the fleet has improved over the time period, and that the method of aggregation used is robust.

Chapter 8

Conclusions and Future Work

8 .1. Summary and Conclusions

The literature review identified three weaknesses in current sustainability assessments of transport systems. These were an incomplete understanding of the concept of sustainable development as it relates to transport, a lack of a clear framework or methodology by which the indicators are selected and a lack of practical application to real systems. As a means of addressing these issues, the Process Analysis Method (PAM) was used to generate a consistent set of indicators describing the sustainability impact of car-based mobility. The process of applying the method identified the key issues surrounding the sustainability of a transport system, which ensured the assessment was both coherent and focused. The indicators were applied to the UK car fleet in 1995 and 2005, both as a practical means of testing the indicator set and as an investigation into how the sustainability of the car fleet has changed over a ten-year period.

The application of the PAM required an analysis of the system and its various stakeholders. The system processes were identified, and analysed in terms of their impact on sustainability. As well as identifying more conventional impacts such as global warming and other pollutant emissions, this analysis identified that the characteristics of the mobility produced by the car fleet have an important impact on sustainability. As a means of capturing the impact of car-based mobility, a set of *service quality* and *equality of modal opportunity* indicators were generated. It is suggested that a consideration of the quality of service provided is crucial to capture the sustainability of a system (transport or otherwise), for several reasons. Firstly, it affects the level of benefit provided by the service, which affects the extent to which the system is meeting human needs as defined by Brundtland. Secondly, it impacts on user choice. The PAM identified that users were important External Impact Receivers, but also that *user decisions* were significant Internal Impact Generators. Service quality thus affects the sustainability of the wider system, and creates a potential feedback loop – service quality drives user choice, which drives system change, which can then affect service quality. An example of this is journey time: low travel times will attract more users, but the increase in users can cause congestion, thus increasing travel time.

There are some limitations to the indicator set that was generated. Firstly, infrastructure was omitted from the system boundary. Aside from the fact that this excludes a number of important environmental and economic impacts due to road construction and maintenance, it also divorces the transport system from the built environment of which it is a part. This means that infrastructure capacity is not included in the analysis although it has a significant impact on service quality. It is also clear that aspects such as urban form are very important in determining transport demand. Secondly, assessment of service quality is difficult as quality can change depending on the specific trip (in terms of time and place) being undertaken, the purpose of that trip (shopping may require more luggage capacity than commuting for instance), and user preferences. The indicators selected were based on issues found to be important by the literature, but may change as new issues emerge or new data sources become available.

The indicator set is, at a top level, general enough to be applicable to any transport system, although the application to the UK car fleet required the use of more car-specific metrics. It is suggested that this indicator set should form the basis for future sustainability assessments. One such question is the impact of electric vehicles, which are currently the popular solution to making car transportation ‘green’. The indicator set suggests that introducing electric vehicles may, at best, have only a limited positive impact. This is for several reasons. Firstly, electric vehicles do not eliminate emissions so much as displace them to power stations (for the foreseeable future). Whilst this could have a beneficial impact on health impacts, by virtue of not emitting pollutants in immediate proximity to highly populated areas, it does not address issues such as global warming. Furthermore, a significant portion of particulates emitted by vehicles actually stem from brake and tyre wear, and are thus not solved by electric propulsion systems. Emissions aside, electric vehicles create issues in other areas of the assessment: they offer no solution to congestion, road safety, or material and energy intensity. Their service quality may in fact be worse than conventional vehicles due to long battery charging times. This example highlights how the indicator set is useful in a wider sustainable transport context. However, performing such an analysis fully would be a significant extension to the current study, and is suggested as an area for future work.

The study of the UK fleet in 1995 and 2005 proved to be an interesting practical application of the indicator set, and yielded good insight into how the sustainability of the fleet has changed in terms of environmental, human/social and economic impacts. Three trends drove changes in the environmental domain:

- Increasing fleet size and total distance driven,
- Increasing vehicle weight,
- Increasingly stringent environmental regulations.

Annual vehicle-km increased by 16%, whilst fleet size increased by almost 29%. Therefore, the per-vehicle mileage reduced, but the large increase in vehicle (and hence user) numbers ensured that traffic still saw a significant increase. The increase in traffic is important as it affects fuel consumption, which can directly be linked to CO_2 emissions and energy resource depletion. Fuel consumption increased over the time period, although not by as much as the increase in traffic would suggest: fuel use per vehicle-km reduced, showing that vehicles are becoming more fuel efficient, but that these improvements are being counteracted by the increases in vehicle-km. Fleet size also affects the amount of new registrations every year (the ratio of fleet to new vehicles remained constant at around 10%). The increase in new vehicle registrations, alongside an increase in average vehicle weight, caused an increase in embodied energy use and emissions due to the vehicle life cycles. The most interesting trend was the effect of regulation. Several pieces of legislation came in force during the study period, namely the EURO 2 and EURO 3 emissions standards, and the European End-of-Life Vehicle Directive. The EURO standards regulated the emission by cars of non-CO_2 pollutants, which saw a drastic reduction over the time period, despite the increase in vehicle-km. The ELV directive set targets for recycling, reuse and recovery, a result of which waste-to-landfill also reduced despite increasing ELV weight and numbers.

Mobility and impact on health also saw changes over the period. Total mobility increased over the period, meaning the car fleet made a greater contribution towards meeting transport needs. Whilst this is a positive impact, it should be noted that this is a clear example of an indicator running against another: greater mileage increases fuel consumption and emissions, as well as traffic, which leads to congestion. On a per-vehicle basis, vehicle-km reduced, suggesting that car owners are benefiting less from their vehicles in terms of quantity of mobility provided.

The picture in terms of service quality was mixed. Evidence showed a drop in average speed and reliability (from both the network and vehicle perspectives) but an increase in comfort and vehicle safety. The reduction in average speed (of around 5%) suggests greater congestion, which is evidence of the capacity of the road infrastructure being reached. Congestion not only affects the service quality but also emissions – driving in congested traffic conditions increases fuel consumption. Increases in comfort also have wider implications, with additional features such as air conditioning and electric seats increasing fuel consumption and adding weight. Issues with vehicle reliability may have contributed to the expansion of the maintenance industry.

Changes in health impacts were broadly positive over the study period: mortality due to air emissions reduced by 18.4%, and casualties from road accidents involving cars reduced by 13%. The change in morbidity due to air emissions is less clear, either reducing by ~5.5% or increasing by ~2% depending on the choice of ERC. Any reduction of health impacts due to air emissions is due to the reduced emissions of pollutants by vehicles, which has already been linked to legislation. The improvement in road safety is due partly to additional safety features being added to new vehicles, which contributed to vehicle weight increases, although it is likely that other factors such as better road management, driver testing and road safety awareness also had an impact. It was shown that the impact from road fatalities was approximately three times larger than the impact from emission related mortality, suggesting that this should remain a priority for policy makers. Nevertheless, the numbers prove that the impact from emissions is by no means trivial. It is worth noting that health impacts also have an impact on the economic store of value, in terms of costs of treatment and lost productivity.

The economic store of value also saw changes. The increase in fleet size and new registrations drove growth in the retail and maintenance sector, which saw growth of 25% in gross value added. This is the opposite of what happened to the manufacturing sector, which saw a reduction in value added of almost 26%. Several factors contributed to this. Firstly, despite the increase in new registrations, domestic production did not increase significantly. Secondly, it is likely that profit margins shrank under the pressure of adding more features (comfort and safety) to new vehicles whilst reducing retail price. Government revenue also increased overall. The small drop in revenue from vehicle excise duty was more than compensated by the increase in revenue from fuel duty, which suggests that government policy changed with regard to taxation on car use. It should however be noted that due to the system boundary excluding infrastructure, government costs related to managing the road system and road maintenance are not considered. It would be interesting to see how these compare to the revenue collected.

A final point, which underlies many of the changes discussed, is population growth. A simple analysis of population data over the period along with data on fleet size shows a very high degree of correlation between the two ($R^2 = 0.995$). Although correlation does not imply causation, in this case it seems reasonable to assume that population growth has been the main driver of the increase in fleet size, which can in turn explain the increase in mileage.

As a means of summarising the changes in the car fleet across the three sustainability domains, an aggregation was performed. This showed that the overall sustainability of the system increased by 13% over the time period. Whilst this is a clear improvement, it is slow. Equally, some headline issues such as CO_2 emissions and congestion are getting worse, despite widespread awareness. In terms of methodology, the indicators were normalised to their percentage change over the study period, after which a weighted sum was performed. The weights were assigned according to a monetary valuation of each indicator in 2005. The choice of percentage change as a unit is appropriate for the sustainability assessment of a system such as the car fleet. Absolute thresholds for sustainability only make sense on a top-level scale. For instance, sustainable levels of greenhouse gas emissions can only be set on a global scale. Even if a greenhouse gas allowance were apportioned to the UK, a question would remain as to how to disaggregate it for different sectors or systems within the UK. These are political and societal questions, and outside the realm of a sustainability assessment. Therefore, in the absence of mandated thresholds for sustainability, it is more appropriate to discuss sustainability in terms of change. The use of monetary valuation for the weights, whilst potentially controversial, was also justified. Weighting is necessary for aggregation, and monetary valuation has been the subject of extensive research. It is thus well understood. The weights produced, although biased towards economic and mobility impacts, yielded reasonable and robust results.

8 .2. Future Work

Several areas were identified for future work. First is the inclusion of infrastructure in the system boundary. Infrastructure is an important aspect of the sustainability of a transport system, affecting resource use and environmental impact due to its construction and maintenance. It also affects the system capacity or maximum load, which in turn impacts upon performance. A congested network is less reliable, slower, and consumes more energy per km. Infrastructure is also the link to urban form, and the transport system's impact on the built environment. This has consequences on quality of life within that environment. Conversely, the built environment affects infrastructure and transport, through the influence it has on demand and the spatial constraints it creates. Including it in the analysis is therefore a natural next step. A useful indicator to this end would be an indicator of *spatial efficiency* of the transport system. It is clear that people tend to concentrate in areas of interest, especially in urban environments. An analysis of the space available to transport infrastructure in a particular area (for instance, by looking at satellite imagery) could be performed and compared to daily concentrations of people in that area and the space requirement per passenger for a given mode. This should make it possible to calculate the maximum fraction of people who could visit the area using that mode. It is likely that GIS data would be necessary.

Another extension of the boundary would be to include the institutional infrastructure. There are a host of organisations involved in administering the UK car fleet and road system, namely, the Department for Transport, the Vehicle Certification Agency (responsible for certifying the emissions characteristics of new vehicles), the Driver and Vehicle Licensing Agency (responsible for managing the fleet and collecting VED) and the Vehicle and Operator Services Agency (responsible for performing M.O.T.s). These influence (to a greater or lesser extent) the car fleet and its sustainability. Indicators for institutional sustainability could be generated. An important aspect for institutions is the collection of data with which to perform transport sustainability assessments.

A natural extension of the study would be to assess different modes or vehicle types. A cross modal assessment would provide a basis for comparison against which to assess the car fleet. Alternatively, multimodal journeys could be considered. This would give a better impression of how well the transport system as a whole is performing. A third option would be to extend the study to new vehicle technologies, such as hybrid, biofuel, or electric vehicles. This would provide an insight into what technologies provide the greatest benefits to sustainability.

An aspect of sustainability that was not considered in this study is the adaptability or flexibility of a system to change. For true, long-term sustainability a system has to be adaptable to changing circumstances, such as increases (or decreases) in demand, scarcity of particular resources, or regulation. Criteria could be selected surrounding the resources used by the system and the ‘stocks’ (vehicles and infrastructure) contained in it. For instance, what would happen if an important energy source (e.g. petrol or diesel) became scarce or unavailable? What alternatives exist? What level of modification would have to made to the vehicle fleet? It is suggested that this assessment could be modeled on procedures such as FMEA (Failure Modes and Effects Analysis). This is a design and development tool that lists potential ways a system could fail, along with ratings for likelihood of occurrence, detection and severity. A similar method could be employed for adaptability. A list of potential shocks would be created, with a likelihood of occurrence and a description of possible responses. These would be rated in terms of feasibility and ease of implementation.

In the aggregation performed in Chapter 7, the weighting of the indicators forming the mobility (including distance, quality and equality of opportunity) component of human/social impacts requires further validation, as it was performed according to the author's opinion. A statistically significant survey should be constructed, controlling for socioeconomic status as well as availability and quality of public transport to discern the preferences of the users and communities on the various indicators of mobility. The AHP could be a useful tool for this survey to assist respondents in revealing their preferences.

Another area of the study that would benefit from future work is the analysis of the impacts of noise on health. An analysis similar to that on the health impacts of air emissions could be performed, with exposure data (proportion of the population subject to noise levels above a certain threshold) being combined with exposure-response coefficients and an estimate of the share of noise emissions due to vehicles to calculate noise-related health impacts due to transport.

A final area of interest is the development of a model to forecast changes into the future. Predicting the impact of current trends such as fleet growth on sustainability can support policy decisions and determine priorities. Equally, how, and at what rate, new regulation or technologies will affect sustainability is useful for policy design and selection. Changes to the vehicle stock occur slowly, due to long design and development times and a fleet renewal rate of 10-15 years. It is suggested that a systems dynamics model would be most appropriate for modeling a system such as the car fleet, due to its ability to easily deal with stocks and flows.

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