Modelling local environmental quality and its impact on health

Background document for an international scientific audit of PBL team LOK

Background Studies

Netherlands Environmental Assessment Agency

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Abstract

Modelling local environmental quality and its impact on health
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This report describes the methods used by the Quality of the Local Environment Team (LOK) for the evaluation of government policy concerning the quality of the physical local environment and the consequences thereof for health and well-being. This report has been written because of an international scientific audit of this part of the LOK programme.

The Advisory Council of the 'Netherlands Environmental Assessment Agency' (PBL in Dutch) has requested that an international audit of the PBL methods be conducted. As part of this PBL-wide audit, the dossiers for the physical local environment of the LOK team will be evaluated. This report has been compiled to provide the audit committee with the information required to conduct the audit. Besides making a scientific evaluation of the work, the audit committee has also been requested to evaluate whether the scientific research links up sufficiently with the policy evaluations. The report therefore briefly describes the relevant policy in the various dossiers, with examples of policy evaluations. This concerns the following dossiers: air traffic, noise, air quality, external safety and health in relation to environmental quality.

Because the policy dossiers are based on different traditions and approaches, the approach to research and policy evaluation also differs between the dossiers. The present report extensively addresses the methods and data that are important to these dossiers and the uncertainties that play a role in them. A separate chapter will describe how uncertainties are dealt with in the policy evaluation.

Key words: audit, physical local environment, health impact, policy evaluation

Contents

1 Introduction

This report describes the methods used by the Quality of the Local Environment Team (LOK) for the dossiers on the physical local environment and health. These methods are being subjected to an international audit. The aim of this report is to provide the audit committee with the necessary information.

Context and aim of the audit

Models play a crucial role in the work of the Netherlands Environmental Assessment Agency. It is impossible to understand the current environmental situation as a whole by means of measurements alone. Still more important is the fact that many evaluations concern future situations and the exploration of policy alternatives, where measurements are impossible by definition.

The mission of the PBL and its practical implementation require that these models and approaches correctly represent the current state of knowledge. Therefore, the Advisory Council of the PBL has requested that an international audit of these methods be conducted. The audit of the work of the LOK is one component of this larger audit.

The most important questions for this audit are the following:
1. Do the models used by the PBL present a relevant reflection of reality?
2. Are the PBL models in accordance with the scientific state-of-the-art?
3. Is the input for the models sufficiently reliable, valid and up-to-date?
4. Are the policy evaluations of PBL/LOK sufficiently supported by the output of the models?
5. Does the PBL deal properly with the uncertainties in the model outputs?
6. In view of the answers to the preceding questions, can the audit committee make any recommendations to the PBL for the future?

The mission of the PBL

The Netherlands Environmental Assessment Agency (PBL in Dutch) supports political and societal decision-making concerning economic, ecological, spatial and social-cultural qualities by evaluating the policy being conducted and exploring future developments, especially with respect to ecological quality.

Implementing the mission

The PBL provides evaluations and explorations of the quality of life in our country in relation to the environmental issue at the European and global scales. We focus primarily on supporting national decision-making on issues involving nature and the environment. We do this in cooperation with other independent planning agencies in the Netherlands and with other research institutions. According to law, the PBL can provide independent advice to the government.

Every year, the PBL publishes reports on the *Environmental Balance* and *Nature Balance*. Every four years, an *Environmental Outlook* and a *Nature Outlook* are published. These publications pay special attention to the possible effects of environmental policy on the health and well-being of people.

As part of this publication process, the Quality of the Local Environment Team (LOK) focuses specifically on local environmental quality and the corresponding impacts on health and well-being.

Readers guide

An international audit committee has been requested to evaluate the model-based activities of the LOK in the context of policy evaluation. This means not only that it is being asked to evaluate the methods (models and data), but also whether the models and the conclusions that emerge from the models are compatible with the context in which they are applied. This report has been drawn up in order to provide the audit committee with the knowledge it requires to conduct the audit. To help the committee evaluate the context of the methods and the policy evaluations, Chapter 2 provides a summary of the policy aspects that play a role in the various dossiers. In this chapter several examples of recently published policy evaluations are provided for each dossier. Chapter 3, with its descriptions of methods and data, forms the core of this report. The understanding of uncertainties plays an important role in research and in the evaluation and communication of research results. Chapter 4 therefore discusses how uncertainties are dealt with in policy evaluation and communication.

2 The LOK dossiers on the physical local environment and health impacts

This chapter describes the dossiers of the Quality of the Local Environment Team (LOK) that concern the physical local environment and the health impacts due to the physical environmental burden. For the dossiers Air Traffic, Noise, Air Quality, External Safety and Health, the chapter describes the most important discussions, the contribution of the PBL to these discussions and how the policy evaluation takes place.

2.1 Air traffic

2.1.1 Developments in air traffic in the Netherlands

Air traffic causes noise annoyance and external safety risks for local residents and it contributes to local air pollution and the global climate issue. The massive increase in air traffic requires continued understanding and options to optimize the balance between the economy and ecology.

The annual *Environmental Balance* always discusses noise annoyance and external safety risks from air traffic in the Netherlands. In addition, the PBL has published three policy evaluation reports about the developments around Schiphol Airport (PBL(a), 2005; PBL(a), 2006 and PBL(b), 2006) and one about the planned policy for regional airports (PBL(b), 2005). The reasons for these separate reports were 1) the evaluation of the Schiphol policy by the Ministries of V&W (Transport and Public Works) and VROM (Housing, Spatial Planning and the Environment), 2) the subsequent Cabinet decision on Schiphol Airport in May 2006 and 3) the change in legislation for regional and military airports (*Regeling Burgerluchthavens en Militaire Luchthavens*, RBML, 2006). Reports PBL(a), 2005 and PBL(a), 2006 not only discuss noise annoyance but also external safety risks from air traffic. Report PBL(a), 2006 discusses the development in air quality around Schiphol. The main conclusions to these studies and the complete reports can be found at http://www.mnp.nl/nl/dossiers/luchtvaart/publicaties/index.html (all in Dutch).

More than 90% of the air traffic (passenger and freight) passes through Schiphol, the national airport. Schiphol is the fourth largest airport in Europe. The air traffic through Schiphol over the past 25 years has increased, on average, by nearly 8% annually. To enable further growth in the future, Schiphol was expanded with a fifth runway in 2003. At present, additional possibilities for expanding air traffic capacity in the Netherlands are being explored.

In addition to the national Schiphol Airport, the Netherlands has four civil airports for larger aircraft and about ten sites for smaller aircraft. There are about eight military airports, some of which can also be used by the general public.

In 2003, 12% of Dutch adults reported that they suffered serious annoyance from the noise of aircraft, 29% reported that they suffered from the noise of road traffic, 22% from noise in their neighbourhood, and 2% from rail traffic noise. The absolute external safety risks involved in air traffic are very low. However, they are relatively high in the Netherlands when compared to the risks of other dangerous activities such as the production and transport of hazardous substances. The noise annoyance near Schiphol Airport is particularly severe. The external safety risks are highest around the regional airports of Maastricht and Rotterdam. The detrimental impact of air traffic on the local air quality is small (< 2%) but is growing fast in absolute and relative terms. International studies show that the contribution of global air traffic to the emissions of greenhouse gases is between 2% to 5%. This contribution is expected to grow rapidly to approximately 10% in 2050.

2.1.2 Air traffic as a subject of policy evaluation

For over 15 years, Dutch air traffic policy has been aimed at giving Schiphol Airport the space it needs to develop into an airport with a large number of intercontinental destinations with connecting flights to the rest of Europe. This is known as the mainport concept. The Dutch government pursues this policy because the Dutch domestic market is deemed too small for a high-quality network with which Dutch passengers and shippers can reach the rest of the world easily and reliably. Such a network requires some of the intercontinental traffic flows to use Schiphol Airport as their stopover to their destination in Europe and vice versa. A large international airport is also attractive for international companies –often cited as a reason in favour of Schiphol expansion.

At the beginning of the 1990s it was decided to expand Schiphol Airport with an additional runway – the fifth runway – in order to allow it to handle the growing volume of air traffic. The were two main preconditions for the fifth runway: the noise annoyance had to be considerably reduced and the external safety risks were not to increase compared to the reference year 1990. This ambitious combination of expansion and improvement of the living conditions in the local environment became known as the dual aim.

Between 1998 and 2002, the current standard for the five runway system at Schiphol Airport was drawn up. The current standard deviates from the standard that was used before and that still applies to regional and small airports. Until 2003, when the fifth runway at Schiphol Airport was opened, the noise standards at Schiphol Airport were maintained on the basis of noise zones. Within these zones, requirements had been laid down for both the noise load and for the built-up zone, i.e. for the existing houses, insulation of houses or the construction of new houses. The idea behind noise zones was to have 'optimal harmony' between air traffic and the buildings on the ground. When the fifth runway came into use in February 2003, the aircraft noise level around the zones was no longer enforced. Since then, there are limit values in L_{den} for the annual aircraft noise on 35 sites around Schiphol Airport and in L_{night} for the nocturnal noise on 25 sites. In addition, there are maximums for the aircraft noise produced in total (Total Volume, also in L_{den} and L_{night}

From 2003 until early 2006, the Upper House of Parliament commissioned research to evaluate whether the aims of the policy had been fulfilled. This research mapped out the developments as they had taken place since 1990 in relation to noise, external safety risks and air traffic emissions (V&W and VROM, 2006). The government also carried out extensive research into the societal significance of Schiphol Airport as a mainport. In May 2006, the main findings of this research and of the evaluation were articulated in the Cabinet standpoint on Schiphol Airport, which continued the earlier theme of air traffic growth with simultaneous improvement of the living conditions in the local environment.

The Cabinet standpoint is currently being elaborated into tangible policy proposals and measures. Increasingly, it is being questioned whether all growth should be accommodated at Schiphol Airport, or whether some growth should be accommodated at one or more regional airports. Currently, the regional airports have limited capacity for taking on any traffic from Schiphol Airport. Moreover, in 2006 it was decided to put the provincial authorities in charge of the policy for these airports. The national government provided a number of rules for this, including several conditions regarding environmental norms and spatial planning. In practice, these rules will lead to standards comparable to those for Schiphol Airport. Meanwhile, it has been decided that the national government will remain in charge of Lelystad Airport.

The policy process in the Netherlands can be characterized as 'corporatistic', which means that a relatively large number of stakeholders participate in the policy process and that a consensus is sought regarding the direction of policy. In addition, a relatively large role is given to several research institutes and planning offices, which can advise and inform the government on the basis of scientific and economic research.

Problem definition and research questions

The characteristics of the policy process with respect to the content and administration are important in the PBL evaluations. On the basis of these characteristics, PBL has defined the problem as follows.

– Is it possible to further optimize the relationship between the economic advantages of an increase in air traffic and the negative effects of this air traffic on the residents near Dutch airports?

This problem definition has been converted into the following research questions:

– What are the negative effects and how can they be assessed and weighed? How can the negative effects be weighed against the positive economic effects?
– Can improvements be achieved mainly through technological innovations of the fleet, innovations in flying procedures (use of runways and routes), better adaptation of the airport to its surroundings (spatial planning), reconfiguration of the airport, rearrangement or expansion of the Dutch aviation infrastructure, or combinations of these factors? What are the advantages and disadvantages of these options?
– How do the various options relate to the broader national and international context? How does the policy for air traffic relate to the policy for other sources of noise annoyance and external safety risks? How do Dutch airports perform in comparison to each other and how does Schiphol Airport perform in comparison to other large European airports?
– What are the possibilities for attuning the norms to the basic principles of the policy?

2.1.3 PBL evaluations of Dutch airport policy

Air traffic in the *Environmental Balance*

Environmental Balance 2005

"The noise annoyance from air traffic from Schiphol Airport has nearly been halved in the last 15 years, and will remain stable until 2010. With the fifth runway, the number of houses falling within the local risk contours was halved and was below the level of 1990. In 2010, the group risk for external safety will approximately double due to more aircraft movements and more offices and industry near the airport.

Environmental Balance 2006

"The policy goals for noise annoyance and safety risks from Schiphol Airport air traffic were attained. However, such goals only apply to the most heavily affected residential areas around the airport. Most annoyance and risks occur in a much larger area around Schiphol Airport, where the noise annoyance has developed less favourably and where the external safety risks have increased.

Environmental Balance 2007
"A considerable reduction in the total number of people suffering from serious annoyance and sleep disturbance can be achieved around Schiphol Airport. If air traffic intensifies further, this will only be possible if the current system of noise norms is reconsidered and/or some of the air traffic is relocated to Lelystad.

The environment around Schiphol 1990-2010; Facts And Figures (August 2005)

Less noise annoyance
Since 1990, the total number of people suffering from noise annoyance caused by Schiphol Airport air traffic has decreased by about 40%. This is mainly due to the use of new, quieter aircraft. The decrease in aircraft noise is not the same everywhere. In some places, there may even be an increase, for instance under new air routes once the fifth runway became operational. The noise norms limit the current annoyance to about 330,000 people suffering serious noise annoyance and 140,000 with serious sleep disturbance.

More safety risks
The number of houses that may be hit by a crashing plane more than once every million years, (houses within the risk contour) has remained about the same due to the fifth runway. However, the risk of a plane crash with several fatalities on the ground (the group risk) is about twice as large as it was in 1990. The group risk has mainly increased because of new offices and houses being built around the airport. In addition, the risk of a plane crash at Schiphol Airport has increased by 30% to 40% since 1990. Flight safety may have improved per flight, but this is completely nullified by the increase in the number of flights.

The fifth runway has little effect on decreasing the noise annoyance
The opening of the fifth runway in 2003 did not contribute much towards decreasing the noise annoyance (5%). However, the fifth runway has decreased the external safety risks considerably. The number of houses within the risk contour of the four runways doubled between 1990 and 2003, but this increase was almost completely counteracted by the fifth runway. From 1990 to 2002, the group risk became three times as large. It is currently about twice as large as in 1990. It is unlikely that the noise annoyance and the risks will decrease due to any further expansion of Schiphol Airport involving additional runways. Any expansion that spreads out the air traffic further will mean even more annoyance.

Relatively little environmental impact compared to other countries
Around Frankfurt, Paris Charles de Gaulle and London Heathrow, the exposure to aircraft noise is 2-10 times as large as that at Schiphol Airport. The extent of the total group risk for Frankfurt and London Heathrow is 30-50 times as large. At Schiphol Airport, the location of the five runways is relatively favourable in relation to the houses.

Nationally, the environmental norm for Schiphol Airport is relatively lenient; there are stricter norms for other sources of noise annoyance and external safety risks. If air traffic were treated according to these stricter norms, this would lead to more restrictions for air traffic or to much larger areas where plans for new construction would have to take the air traffic into account.

For now, noise standards will determine the growth of Schiphol Airport
It is not the physical capacity of the runways or the external safety norm, but the noise standards that will most determine Schiphol Airport's capacity for growth in the next 10 to 15 years. In

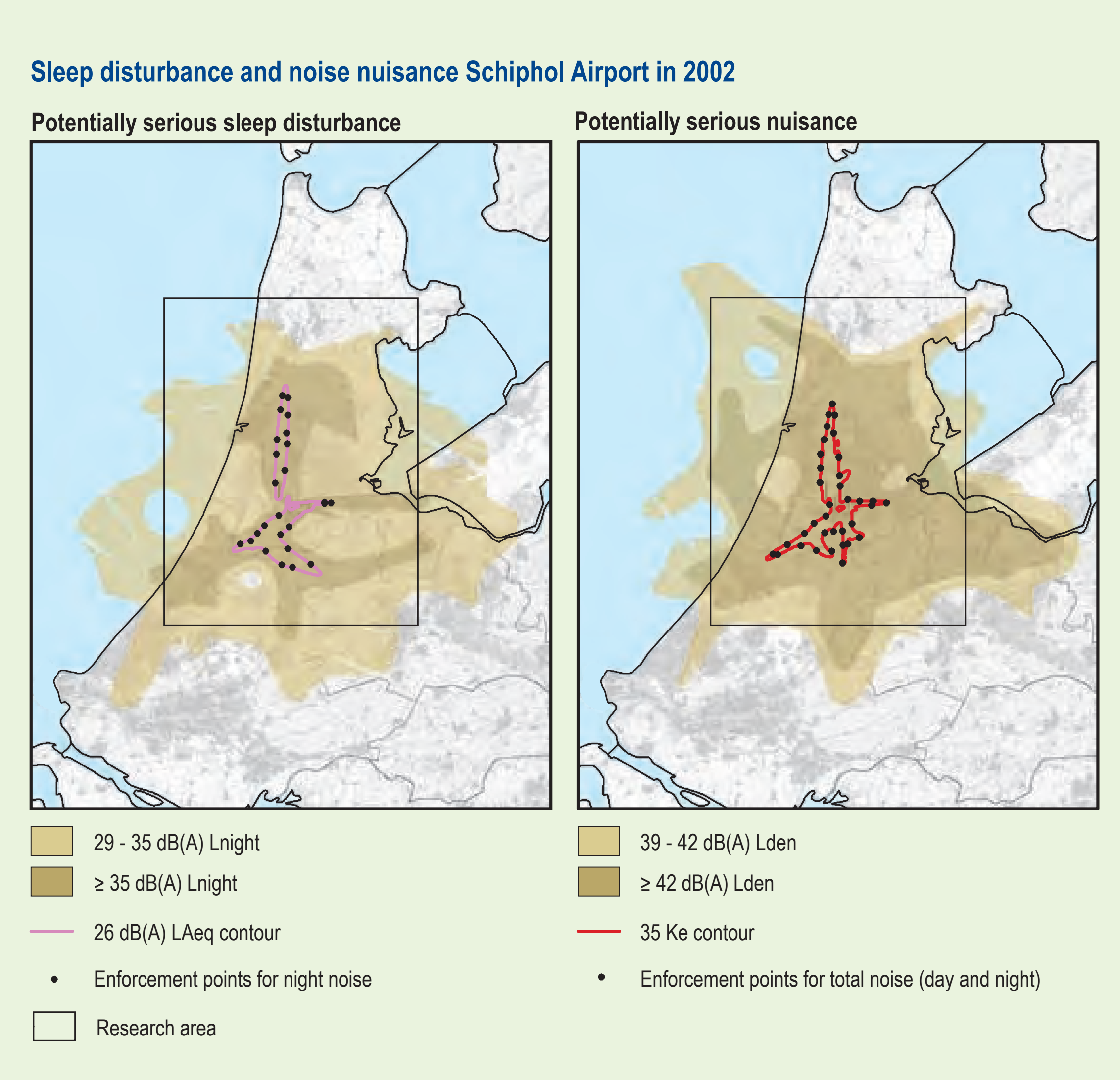

Figure 2.1 Enforcement points for noise (total noise (day and night) on the left, night noise on the right) and the areas which are affected by noise annoyance (on the left) and sleep disturbance (on the right). The map shows the contours of the inner area (within the enforcement points) and the outer area where most people who experience noise annoyance live.

2010, between 480,000 and 550,000 flights can be handled within the noise norms. However, in order to reach this target, the airport would have to handle a limited number of the noisiest flights during the day instead of during the late evening or night. About 420,000 flights are currently being handled (2005). After 2010, capacity for growth will greatly depend on the air fleet becoming quieter.

Options for Schiphol policy, balance between inner and outer area (June, 2006)

It is not possible to simultaneously let Schiphol Airport grow, stabilize the noise annoyance and risks in the inner area, and reduce the noise annoyance in the outer area. Earlier investigation showed that considerable growth of the air traffic within the current norms for Schiphol Airport is possible (see Het milieu rond Schiphol 1990-2010 - Feiten en Cijfers, PBL 2005 (The Environment around Schiphol Airport 1990-2010 – Facts And Figures)). However, a simultaneous substantial reduction of the noise annoyance in the outer area is not possible.

Substantial reduction of annoyance is possible
If an effective policy is implemented for the outer area (see Figure below), it is possible to decrease the number of people that are suffering serious noise annoyance by 45,000 and the number of people suffering serious sleep disturbance by 60,000, assuming growth in aircraft movement from 420,000 to over 600,000 in 2020. This is also beneficial for the extent of the external safety risks, the required space and the environmental costs of the air traffic. The price to be paid is a net increase of a few thousand houses with a high noise level in the inner area. The noise level and air traffic risks are highest in this area, but only 2% to 3% of the local residents experiencing noise annoyance live there.

Environmental costs increase when air traffic increases
An increase in the environmental burden from air traffic causes additional health effects and risks. This could result in increased costs for housing insulation, loss of space and lost value of houses; these aspects may no longer weigh against the advantages of extra travel options for the Dutch consumer. An increasingly strict norm for the outer area can result in maximum use of innovations and fewer flights over more densely populated areas.

The proposed trade-off is not effective
The trade-off proposed by the Cabinet, which maintains the current protection of the inner area, does not offer the intended growth for air traffic. Adapting the limit values to the enforcement points will be effective only if it is made subsidiary to improvements to the outer area.

Explanation of Figure 2.1, Noise annoyance and sleep disturbance at Schiphol Airport
Based on annoyance research around Schiphol Airport, it can be expected that when local residents are exposed to 39 dB(A) L_{den} (total noise, day and night) and 29 dB(A) L_{night} (night noise) about 5% of the people will still experience serious annoyance from aircraft noise and 3% will experience sleep disturbance. Other European countries assume threshold values for such effects of 42 dB(A) L_{den} and 35 dB(A) L_{night}. Within the ring of enforcement points, the maximum noise level is limited. The ring of enforcement points contains about 3% of the total number of people who experience serious noise annoyance, and 2% who experience sleep disturbance. Outside the ring of enforcement points the noise level decreases, but the actual noise level may vary greatly.

The noise contours are based on the air traffic in 2002. The report, Het milieu rond Schiphol, 1990-2010 - Feiten en cijfers (PBL, 2005), (The Environment Around Schiphol Airport 1990-2010 – Facts And Figures), maps out the realised and expected development in noise annoyance and the external safety risks for the period 1990 to 2010, within the research area.

2.2 Noise

2.2.1 Developments concerning noise levels in the Netherlands

Environmental noise can be a annoyance and lead to sleep disturbance. Surveys have shown that it is especially noise from road traffic, neighbours, air traffic, railroads and industry that can be a annoyance. In the Netherlands, there is an extensive noise policy applying to traffic noise and industrial noise. For the purposes of this audit, the description of noise will focus primarily on noise from road traffic and rail traffic. For these sources, the research was conducted entirely by the PBL with the aid of the EMPARA model. For air traffic and industrial noise, data from third parties was used.

Environmental noise is addressed regularly in the annual *Environmental Balance* of the PBL. In recent years, no separate reports have been published with evaluations of noise policy. However, the recent *Sustainability Outlook* titled *Nederland– Later* examined the expected developments in road traffic noise. The conclusions of the PBL in the *Environmental Balance* publications concern the development of the noise level that affects houses and noise-sensitive areas. But not many conclusions were made about the effects of noise. This is because the policy focuses on reducing the noise exposure of houses and noise-sensitive areas.

The following section provides a brief sketch of the noise abatement policy in the Netherlands. The section after that contains examples of recent data and conclusions from the PBL about environmental noise.

2.2.2 Noise as a topic for policy evaluation

The first legislation for noise abatement went into force in the 1970s. The Noise Abatement Act (1979) and the Aviation (Schiphol Airport) Act (1978) included limit values for maximum exposure to noise from road traffic and air traffic near airports. Following the enactment of this legislation, there were many supplementations and amendments (including limit values for noise from industry and railroads).

At the end of the 1980s, the government established the objective of preventing increased annoyance due to environmental noise and even to reduce serious noise annoyance to a negligible level by 2010. At the beginning of the 21st century, however, this objective was abandoned. It turned out to be unfeasible due to the low sound levels at which noise annoyance occurs. Instead, the fourth National Environmental Policy Plan (NMP4) emphasized 'acoustic quality'. The national government did not provide a substantive definition of this concept. The basic idea is that acoustic quality depends on the character of a specific area. However, as part of the national infrastructure, the national government did establish a concrete target for houses with a relatively high noise level as part of the plans for national infrastructure; these are referred to as hotspots. This concerns houses near motorways with a noise level of 65 dB(A) Lden and higher and houses near railroads with a noise level of 70 dB(A) and higher.

A list was also drawn up of houses with high noise levels (>60 dB(A) not only near motorways, but also near secondary roads and in the inner cities. The intention is to improve the noise situation of the houses that appear on this list. For eliminating the hotspots and providing noise abatement for houses, a large budget has been reserved for the period until 2020 (approximately €1 billion). For the actual noise abatement, the national government relies primarily on initiatives from provincial and municipal governments.

At this time, the Noise Abatement Act is being amended. This will lead to phased modifications in the legislation for road, railroad and industrial noise. As part of the current phase, the system of legislation is being assessed. To promote source control measures, an enforcement method using ceiling values for the noise emission of the sources is being considered.

The national government wants to promote the development and implementation of noise source policy. To this end, a noise abatement innovation programme was established several years ago. This programme was recently terminated.

In addition, the EU Directive on Environmental Noise and the Noise Action Plans are seen as instruments that must promote initiatives for noise abatement.

The problem formulation used by the PBL when evaluating noise policy is the following: “Is it possible to achieve the noise abatement objectives with the strategy that the Dutch government plans on using?”

The problem formulation can be converted into the following research questions:
– What are the options for achieving the noise abatement objectives?
– What are the costs and benefits of the Dutch policy (expressed in not only in monetary terms, but also in terms of the effects on health and well-being)?
– At which levels of scale should noise policy be given shape and implemented? What have been the benefits until now of EU policy (source-based and otherwise) as well as national and local policy?
– What are the options to optimally link up the norms with the basic principles of the policy?
– How does the Netherlands perform compared to other countries?

2.2.3 PBL evaluations of noise policy in the Netherlands

In recent years, the publications of the PBL have included indicators with various noise level standards. This concerns different 24-hour noise weighting methods. There are two reasons for these differences. First, in connection with policy, various noise standards are used for various indicators. Second, the indicator for the exposure of houses to road traffic noise has been recently changed from L_{24h} to L_{den}.

The following are the main conclusions about noise from the last three editions of the *Environmental Balance*:

Environmental Balance 2007:
Due to noise abatement measures on motorways and railroads, the number of houses with high noise levels has recently decreased. In contrast, the total area with low noise levels has shrunk during the last 20 years due to increased road traffic and the expansion of the road network.

Environmental Balance 2006:
Most noise hotspots are the result of municipal roads.

Environmental Balance 2006:
The new indicator L_{den} is less sensitive to noise at night.

Environmental Balance 2005:
For eliminating the noise hotspots along motorways in 2020, source policy, focusing on aspects such as quieter tyres and road surfaces, is more cost effective than installing noise barriers.

Included below is an example from the Environmental Balance 2007; it shows the historical progression of the number of houses with a noise level higher than 65 Lden due to motorways and higher than 70 dB Lden due to railroads.

The development as shown for national highways in the above figure may be too optimistic because acoustic properties of road surfaces may be less effective than previously assumed .

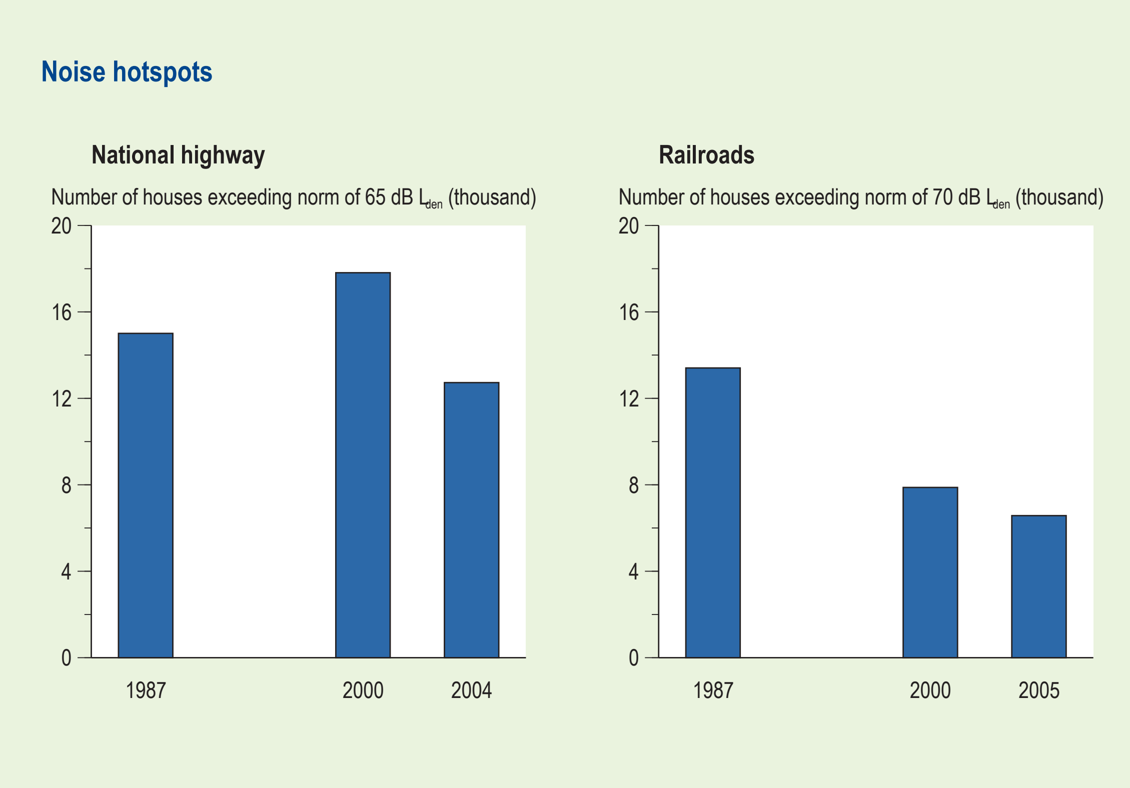

Figure 2.2 Development of noise hotspots near national highways and railroads, from the *Environmental Balance* 2007.

For open graded asphaltic mixes, the national calculation method assumes a noise reduction effect of approximately 4 dB on motorways. However, recent research has shown that the actual reduction is less (M+P, 2007). The average noise-abatement effect of open graded asphaltic mixes decreases as the road surface ages, but there are also large differences between individual road sections. Taking account of the age of the road surfaces, the average noise-abatement effect on the major road network is estimated to be 3 dB, with an uncertainty margin of ±1 dB. As a result, the number of houses near motorways with a noise level higher than 65 dB Lden rises to nearly 13,000, with an uncertainty margin of ± 3,000). The Directorate for Public Works and Water Management is conducting additional research into the acoustic properties of open graded asphaltic mixes; at the same time, the possibilities of efficiently anticipating the results are being studied.

In the *Environmental Balance* 2006, the number of future hotspots is shown on the map of the Netherlands.

Because noise annoyance is not limited to houses with the highest noise level, the PBL has also published a document showing the exposure of houses to noise in a different fashion (as an extension of the formal concrete policy objective). The figure below includes not only national highways as a noise source, but also shows the exposure of houses to all road traffic according to individual road type, as well as the cumulative noise level.

The *Sustainability Outlook 2 (Nederland – later)* showed the effect of the development in the number of houses on this indicator.

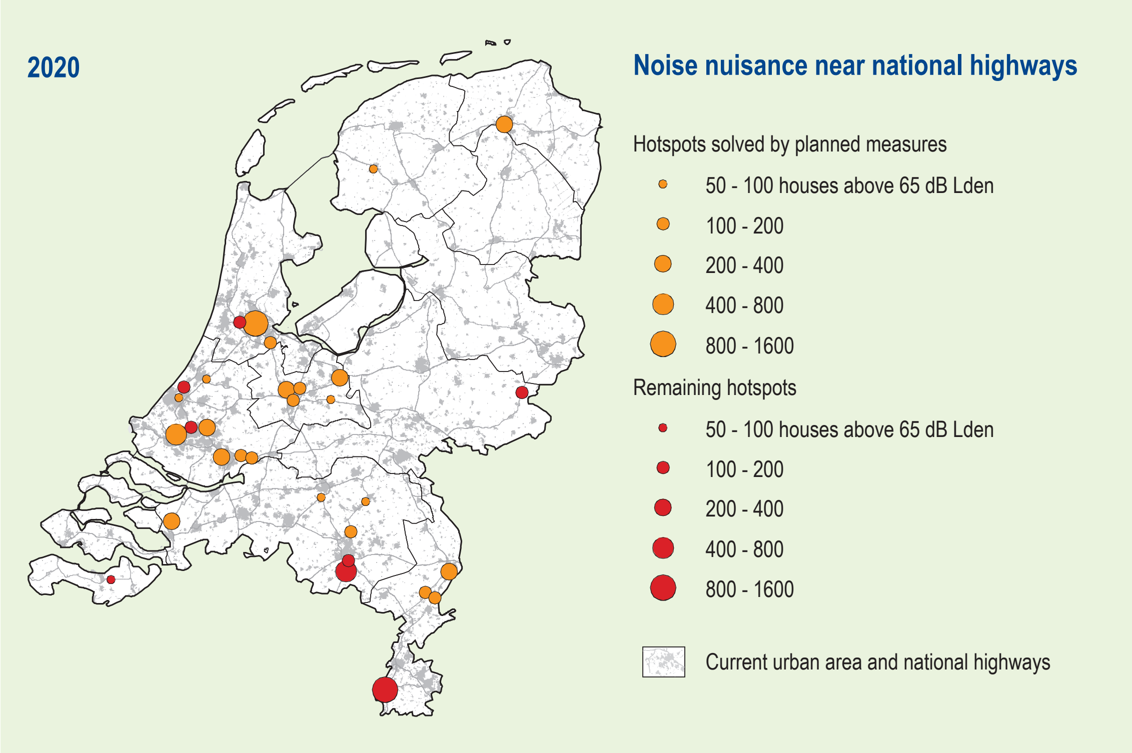

Figure 2.3 Estimation of noise hotspots near motorways in 2020.

Besides the noise level on houses, the area with a specific noise level or a higher was also used as an indicator. The example below shows the results for individual provinces.

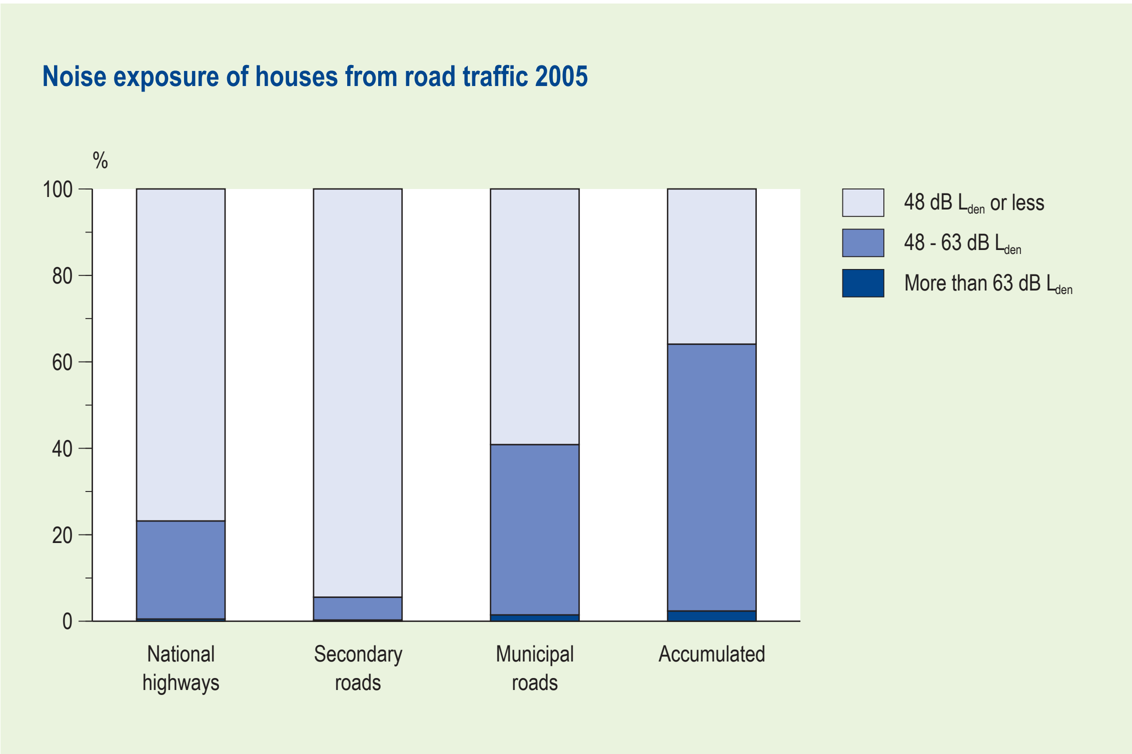

Figure 2.4 Noise exposure of houses from different kinds of roads and the accumulated exposure, 2005.

Figure 2.5 Effect of development in the number of houses on the indicator "noise exposure of houses".

The noise level area is also shown on maps. In the example below, the quality of noise abatement areas and nature reserves is shown.

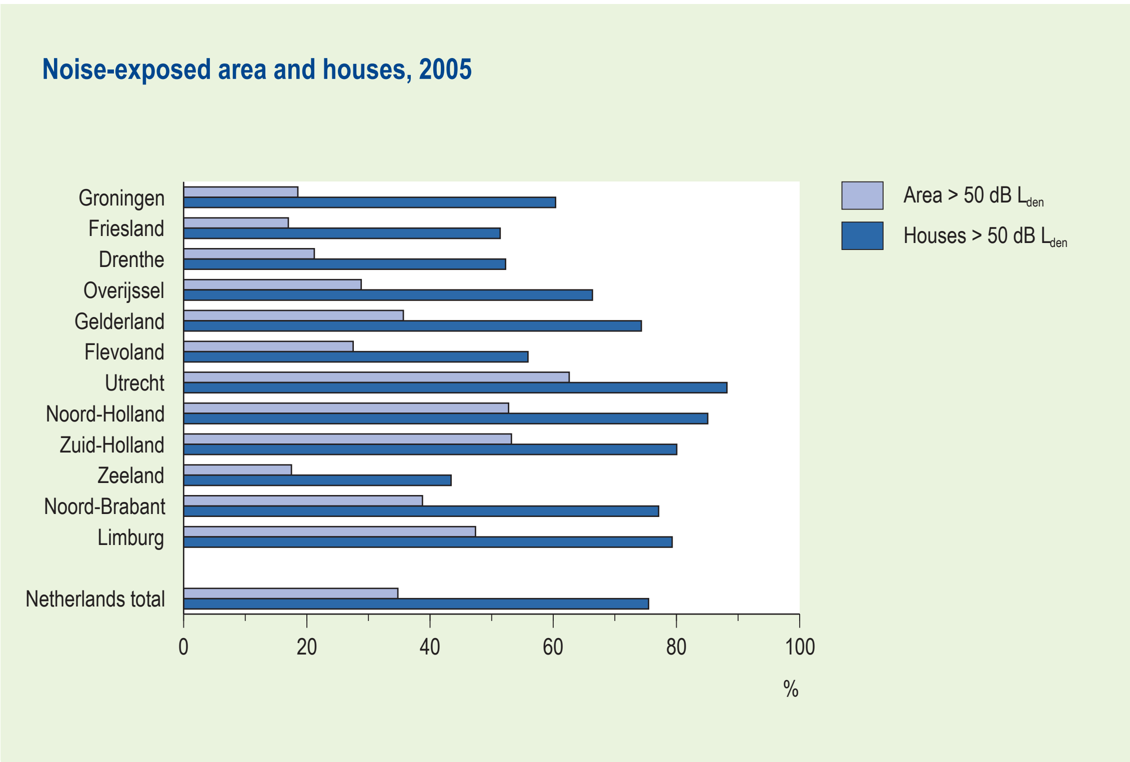

Figure 2.6 Noise-exposed area and houses, differentiated to provinces and the total for the Netherlands. (Source: NLR, AEA Technology, AVV, provincial governments.)

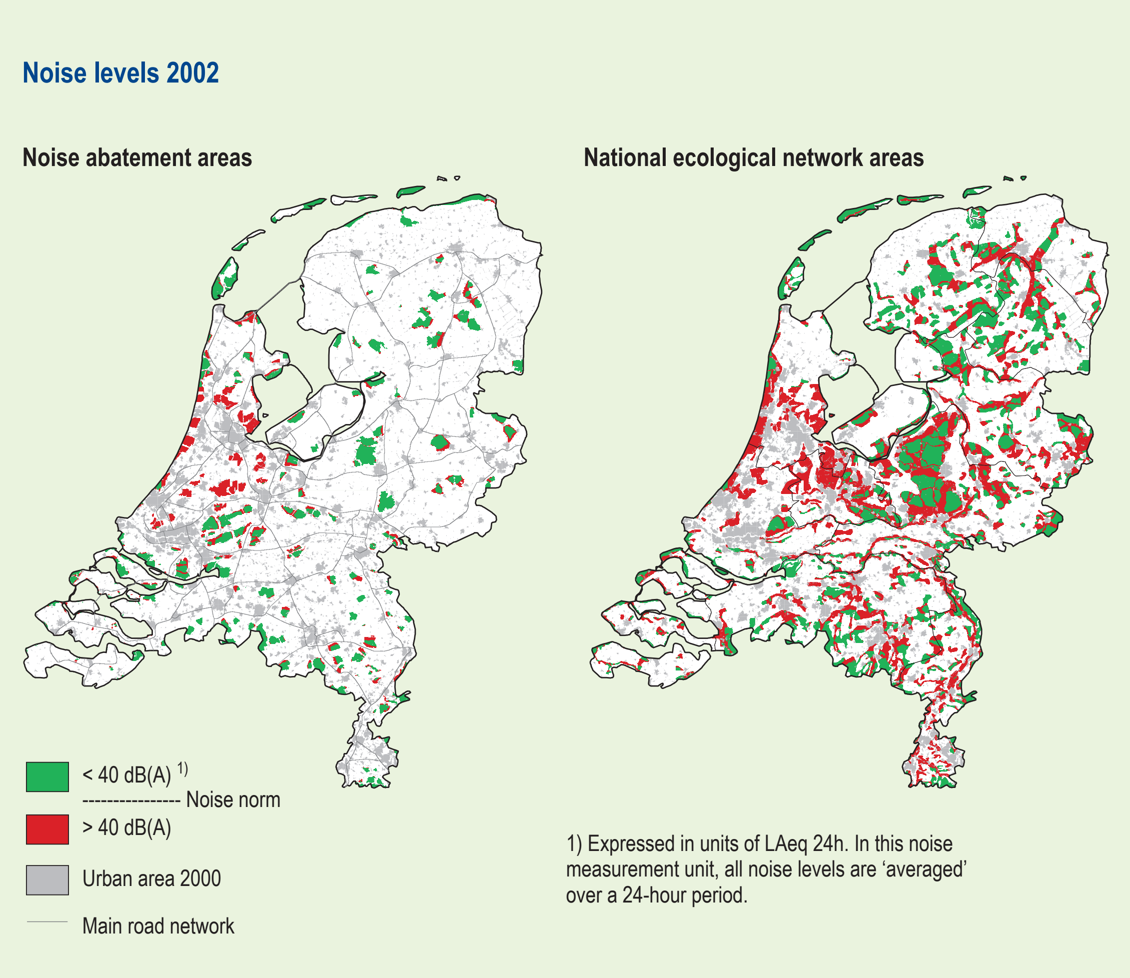

Figure 2.7 Noise levels in 2002, in noise abatement areas and national ecological network areas. (Source: RIVM, NLR, AVV, AEA Technology, Alterra.)

For several cases in the past, the development of noise is shown by means of an average noise level.

2.3 Air quality

2.3.1 Developments concerning air quality in the Netherlands

In recent decades, air quality has improved greatly. Nevertheless, further improvement is necessary because the air quality does not comply with all standards. However, even if the standards are complied with, detrimental health effects will still occur, especially due to particulate matter. This component does not have a 'no-effect level'. Air quality policy focuses on the norms for NO_2 and PM_{10} (the coarser fraction of particulate matter) because for these components, the norms are currently being exceeded. In the near future – when the new European air-quality directive goes into force – there will also be a norm for $PM_{2.5}$ (the finer fraction of particulate matter). The limit values for PM_{10} went into force in 2005; for NO_2 the limit values will go into force in 2010. The limit values for PM_{10} are being exceeded, and those for NO_2 will also be exceeded in 2010. In the current situation, this occurs primarily in busy streets in large cities and along motorways. In the Netherlands, one of the consequences of the

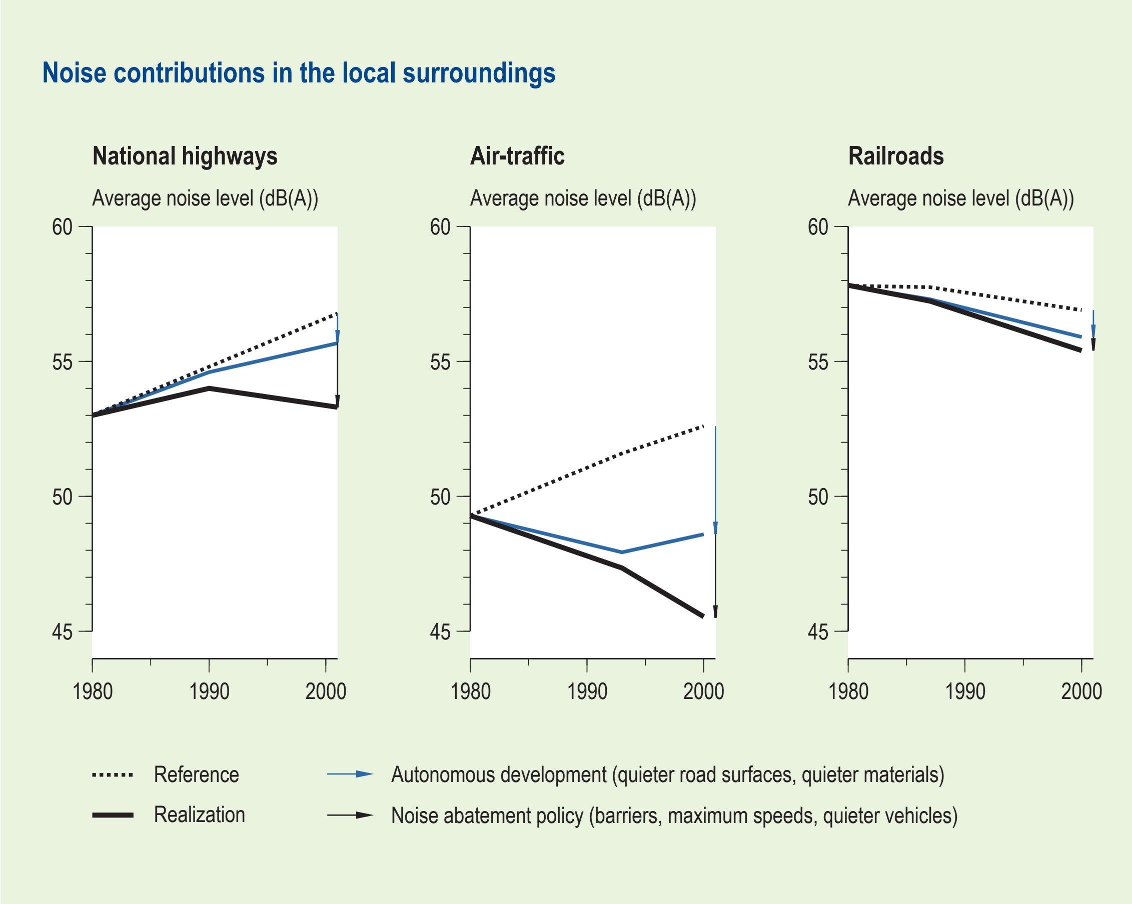

Figure 2.8 Noise contributions of national highways, air-traffic and railroads in the local surroundings.

norm exceedences (expected or actual) is that construction programmes for infrastructure and projects involving land-use planning (business estates, new housing projects) will be halted based on legal grounds.

The new European directive offers the possibility for postponing the date when the limit values go into force (derogation). The Netherlands has requested such a postponement and expects approval because it has shown that the Dutch government is seriously trying to comply with the norms. This means that under the new regime, the limit values for PM_{10} will go into force in 2011 and those for NO_2 in 2015. For $PM_{2.5}$ the limit values will go into force in 2015. Without supplementary policy at the European, national and local levels, exceedences of the norms will still occur after the postponement. Therefore, major efforts at all policy levels are still required to comply with the norms in a timely fashion.

2.3.2 Air quality as a topic for policy evaluation

The Netherlands' air quality policy is multifaceted and focuses on both national and international measures. In addition, legislation is being assessed and the government pays attention to the implementation of air-quality policy at all policy levels.

The national measures focus primarily on reducing emissions from traffic, industry and agriculture.

The Netherlands cannot comply with the norms without major efforts at the European level. Therefore, the Netherlands is arguing especially in favour of stricter European norms regarding emissions from passenger cars, trucks and boats.

Amendments to legislation should make it possible to deal more effectively with the problems related to air quality. For example, it will become possible to make a more flexible trade-off between spatial development and air quality.

Within the Netherlands, all policy levels (national, regional and local) must work together in order to comply with the norms in a timely fashion. In order to give optimal shape to this form of cooperation, the government has established the National Air Quality Cooperation Programme *(Nationaal Samenwerkingsprogramma Luchtkwaliteit – NSL)*. The NSL combines and coordinates all programmes that aim to improve air quality.

Problem definition and research questions
For the purposes of policy, it is important to know the effects of measures at the European, national and local scales. The question here is whether the measures that have been taken or are planned are adequate to comply with the norm; if not, what extra measures are required?

For the PBL, this problem definition leads to the following research questions:
– What are the effects of the measures that have been taken and are planned?
– How do the measures affect the number/magnitude of locations with norm exceedences?
– How can measures be applied optimally in the combination of European, national and local levels?
– What are the costs and benefits of Dutch policy?

2.3.3 PBL evaluations of air-quality policy in the Netherlands

Within the PBL, the team LED (European Air Quality and Sustainability Team) calculates the effects of European and national measures on the large-scale air quality in the Netherlands (large-scale basis map). The team LOK uses Luvotool to provide supplementary calculations about the effects of these measures at the local scale. This concerns only the addition of local effects due to road traffic. Luvotool is not suitable for calculating the effects of specific local measures. Based on Luvotool, statements are made for the past and the future about the following: the areas where norm exceedences occur, the possible level of the concentrations and the number kilometres of roads in cities and along motorways where exceedances of the norm for air quality occur. These statements are based on the air pollution map of the Netherlands; this is the large-scale basis map with the addition of the map showing the contributions of road traffic, which is calculated by Luvotool (see Figure 2.9. Concentration map for PM_{10}, yearly average concentrations for 2006. Detail: area around Rotterdam).

Number of hotspots declines sharply with current policy
(PBL, 2006, available only in Dutch)

With current policy, the number of particulate matter hotspots will decline by more than 50% between 2005 and 2010 (Figure 2.10). Until 2010, the number of hotspots for nitrogen dioxide will decline by around 20% on motorways and by 50% on urban streets. Some policy measures also have an effect over the longer term. As a result, the number of hotspots will continue to decline between 2010 and 2015. The number of hotspots for particulate matter and nitrogen

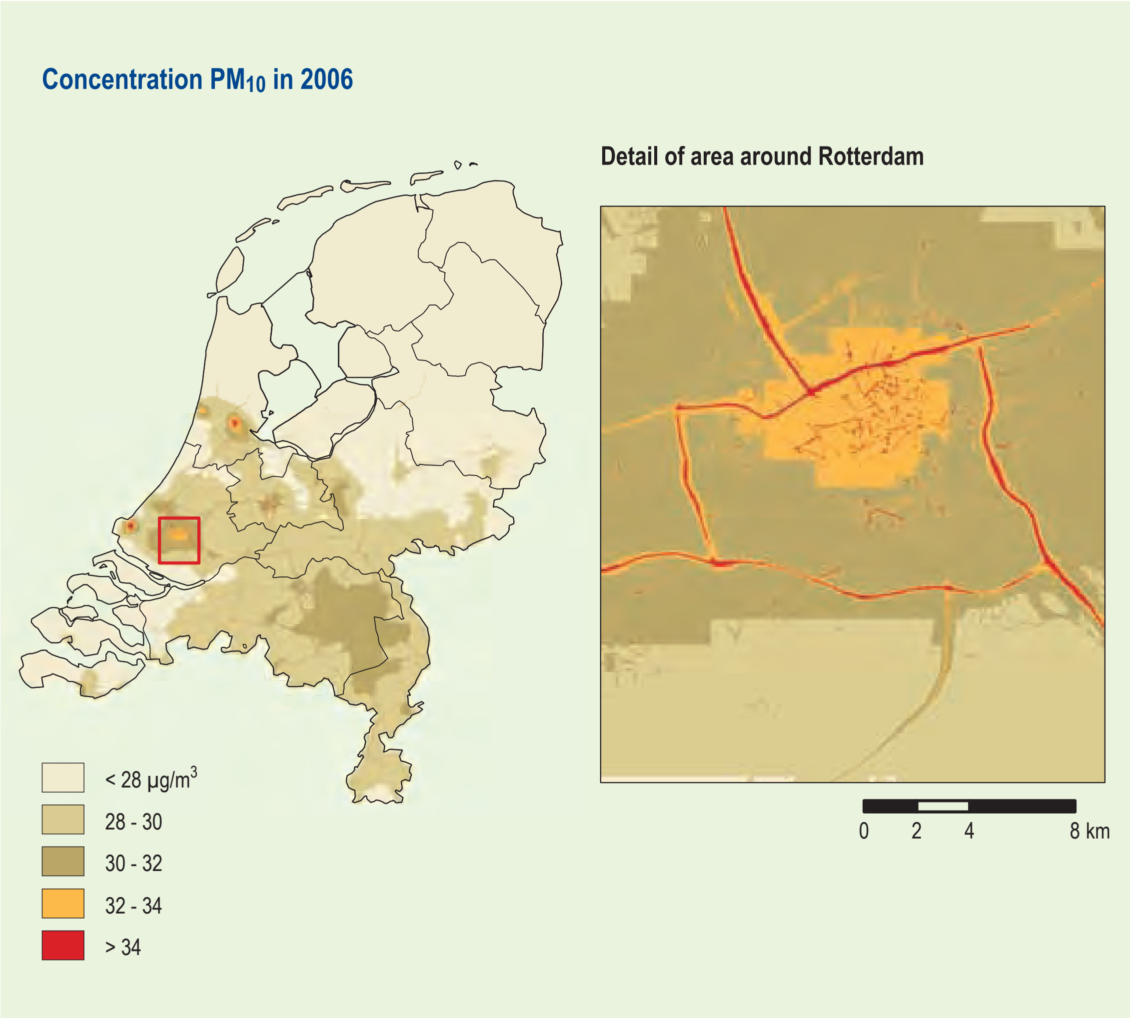

Figure 2.9 Concentration map for PM_{10}, yearly average concentrations for 2006. Detail: area around Rotterdam.

dioxide, assuming current policy, will stabilize after 2015. With current policy, particulate matter hotspots are still expected to occur along busy streets in the big cities in the Randstad and in the southern part of the Netherlands. After 2010, with current policy, the particulate matter hotspots are expected to become concentrated around large cities with large amounts of traffic and industrial activity, such as Amsterdam and Rotterdam.

With current policy, it is unlikely that the annual limit value for nitrogen dioxide will be complied with everywhere in the Netherlands in 2010 and 2015. With supplementary European, national and local policy, it could be possible to eliminate the remaining hotspots for nitrogen dioxide around the year 2015. However, the current proposal of the European Commission for more stringent requirements on the NOx emission of diesel passenger cars does not go as far as was assumed in the Thematic Strategy. With the less stringent requirements that are now proposed, it will possible to eliminate all hotspots with local policy only after 2015.

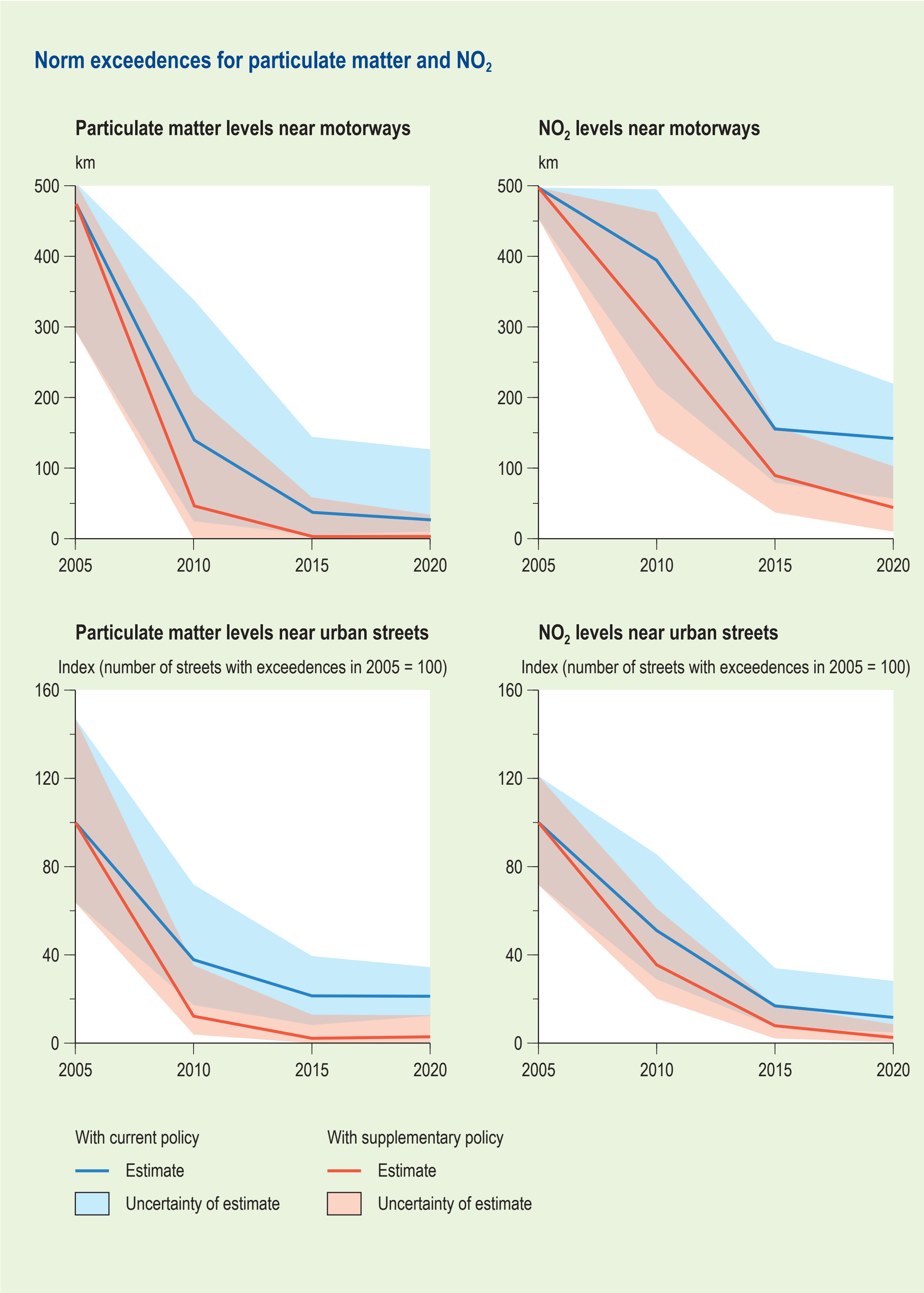

Figure 2.10 The number of hotspots for particulate matter and nitrogen dioxide with current policy and supplementary policy. Topmost figures: kilometres of motorways with exceedances. Lowermost figures: number of urban streets with exceedances. For ascertaining particulate matter hotspots, the contribution of sea salt is not included, in accordance with the Netherlands Air Quality Decree 2005.

Table 2.1 Estimate of the NO_2 concentration near motorways in 2020 with current policy (both excluding and including the recently approved Euro6 NO_x emission standards for light vehicles) and for the two policy extremes 'national emission policy' and 'European emission policy'. The calculations show the average values for a small number of the most heavily-polluted sections of motorways near Amsterdam and Rotterdam.

		excluding Euro6 NO_x norms light vehicles	including Euro6 NO_x norms light vehicles	National emission policy	European emission policy
NO_2 concentration 2020	µg/m³	42.6	39.5	37.2	33.0
Concentration reduction 2020	µg/m³	-	3.1	5.4	9.6
Costs 2020	€M/yr	-	130	280	470

Source: Smeets et al., 2007 (in Dutch)

Environmental Balance 2007
(PBL, 2007, available only in Dutch)

With current policy, air quality will continue to improve during the next 10 years, and the number of air-quality hotspots will decline sharply. However, under current policy it will be impossible to comply with EU limit values at all locations during the period until 2015, even with the options for postponing the application of limit values offered by the new EU air-quality directive.

It cannot yet be ascertained whether this will be possible after the implementation of approved policy, including the regional and local measures that are envisioned in the National Air Quality Cooperation Programme (NSL); this is partly because it is still unclear which measures will be implemented in the NSL and what their effects will be. Moreover, there are uncertainties in the economic and technological developments, as well as uncertainties regarding the dispersal of air pollutants. The ring roads of the major cities and the busiest urban streets in the Randstad are some of the most stubborn air-quality hotspots.

Cost-effectiveness of supplementary measures for cleaner air
(Smeets et al., 2007, available only in Dutch)

Until 2015, air quality in the Netherlands regarding particulate matter (PM_{10}) and nitrogen dioxide (NO_2) can be primarily improved by taking a number of supplementary national and local measures. Such measures can be implemented relatively quickly. This contrasts with more far-reaching European emissions policies, such as source measures and emission ceilings, which require a long time to achieve a full effect. These supplementary national measures are the following: implementation of road-use pricing, technical measures for the storage and handling of bulk goods, soot filters and NO_x technology in inland shipping, advanced dust abatement technologies in industry and air scrubbers on larger pig and poultry housing units. In view of the reduction of the exposure of the population to particulate matter, these national measures are more cost effective than the implementation of more stringent European emission norms (Euro6/VI) for road vehicles. Table 2.1 shows an example of the results from this study.

Benchmark with other countries

Still no uniform approach to external safety policy in EU
In general, it can be said that there are many different approaches to external safety within the EU, and different parties interpret concepts in different ways. This means that a uniform EU-wide approach is an unrealistic ambition at present. In the United Kingdom and Germany, as in the Netherlands, a policy on external safety has been introduced that addresses the various aspects of external safely.

European Union policy on external safety is determined by the Seveso II Directive (96/82/EC) from 1996. The original Seveso Directive (1982) focused mainly on the technical aspects of safety, but over time it has become clear that organization and communication are also essential aspects in terms of preventing major accidents and incidents. The Seveso II Directive therefore requires the companies to which it applies to provide information on the technical and organizational measures they have in place to prevent major accidents and minimize the consequences for people and the environment.

Substantial differences in computing models and methodologies used in different countries
In a number of countries, external safety is an extension of internal safety. The risk source is a specific activity (companies that work with hazardous substances), and requirements are therefore placed on the responsible party via the permit system.

There are significant differences between the use of computing models and methodologies and their outcomes:
- Effect-oriented approach: Germany, Sweden and USA
- Probability approach: Italy, France, Switzerland and Wallonia
- Risk approach: the Netherlands, Flanders and United Kingdom

Spatial planning plays a role in all countries neighbouring the Netherlands. In many cases, companies with a high external-safety risk are concentrated in large industrial zones surrounded by safety zones within which, for example, no vulnerable objects must be located.

2.4 External safety

2.4.1 Developments relating to External Safety in the Netherlands

Statutory norm for location-related Risk not greater than once in 1 million years
In 2004, the External Safety (Establishments) Decree (BEVI, *Besluit Externe Veiligheid voor Inrichtingen*) created a link between spatial planning and the environment. The BEVI contains hard targets for realizing a minimum level of protection for citizens. At locations with houses or other vulnerable objects, by 2010 the annual risk of dying as a result of an incident with hazardous substances must not be greater than once in 1 million years ($PR=1\times10^{-6}$).

Group risk: the probability of a disaster involving many people
There is no statutory norm for group risk as there is for location-related risk. For group risk there is an orientation value. Public authorities must specify reasons for allowing activities that deviate from this value. In making such decisions, they must consider factors such as self-reliance and controllability.

Norms for airports are more flexible
A location-related risk above 10^{-6}/year and up to 10^{-5}/year is accepted around airports with 5,000 to 7,000 individuals living in the vicinity. This is 10 times higher than the permitted norm for other activities. Around the regional airports, there are currently no restrictions on office construction. If these airports become the responsibility of the provincial authorities in 2008, the authorities can formulate their own criteria for this.

Transport network for hazardous substances
The Ministry for Transport, Public Works and Water Management (V&W) is preparing to introduce road, rail and water transport networks for hazardous substances (V&W, 2005). In 2010, if this intended policy is implemented, certain routes will be subject to transport restrictions and others to spatial planning restrictions. Some routes will be subject to both types of restriction.

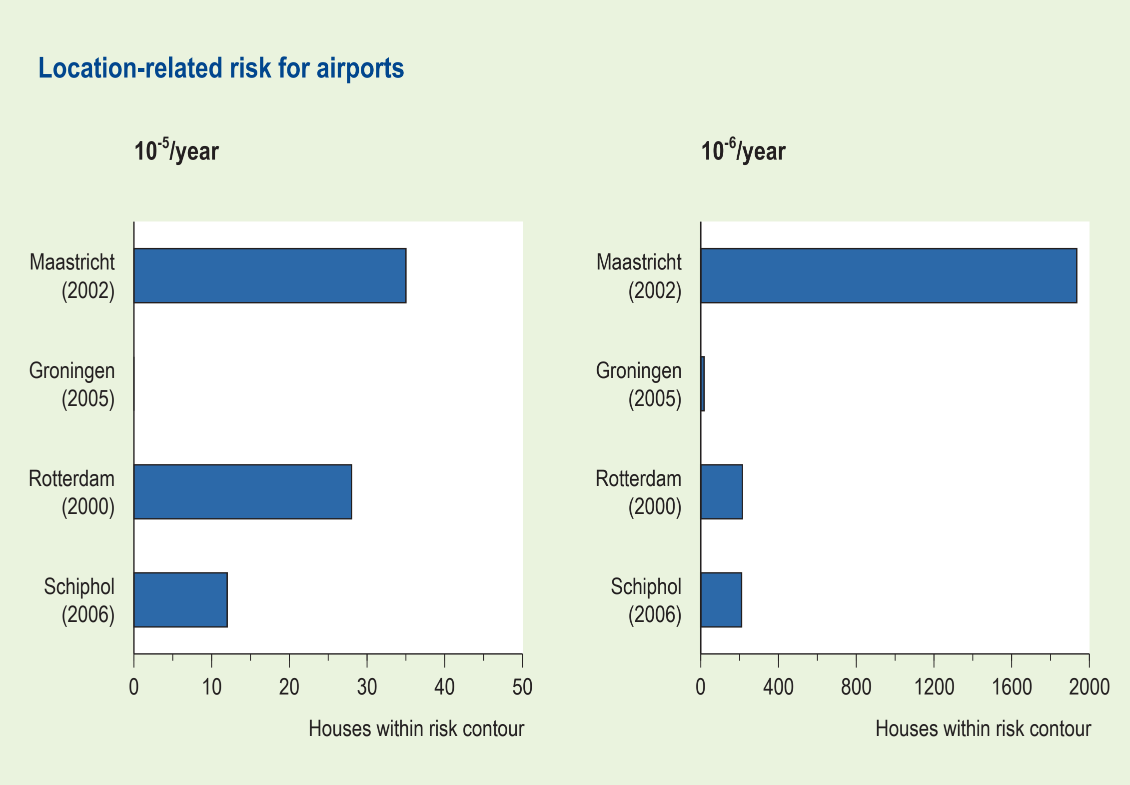

Figure 2.11 Houses within 10^{-5}/yr and 10^{-6}/yr location-related risk contour around airports.

2.4.2 PBL evaluations of External Safety policy in the Netherlands

More houses within PR 10^{-6} contour around regional airports than around Schiphol
In 2006, approximately 200 houses around Schiphol were subject to a location-related risk greater than 10^{-6}/yr. This was nearly 40 fewer than in 2005 and approximately 1,400 fewer than in 2002, before the fifth runway was built. In 2006, fewer than 20 houses were subject to a risk greater than 10^{-5}/yr. Under the present norm for external safety, up to 900 houses around Schiphol could be exposed to a location-related risk greater than 10^{-6}/year (PBL, 2005b), and a few dozen to a risk greater than 10^{-5}/year. Around the regional civil airports, this risk figure is approximately 2,200 and 60 houses, respectively. The largest number are located around Maastricht Airport (Figure 2.11), where built-up areas are relatively close to the runways.

Group risk for Maastricht comparable to that for Schiphol
The group risk for Maastricht Airport is comparable to that for Schiphol (Figure 2.12 and Figure 2.13). The relatively high external-safety risk at regional airports is mainly due to the fact that they are located close to built-up areas. For all regional airports combined, the total number of houses within risk contours *and* the risk of an aviation accident are therefore greater than for Schiphol. The external safety risk within the existing environmental space may increase considerably as a result of planned construction around regional airports (PBL, 2005b). In 2006, the group risk around Schiphol was approximately half of the maximum permitted risk for the external-safety norm. In 2006, the total group risk around Schiphol was almost equal to the group risk in 2005.

In a densely populated country like the Netherlands, people are exposed to a certain level of risk from dangerous activities such as the use, storage and transport of hazardous substances, or risks from air traffic. These risks are classified under the heading 'External Safety' (known by

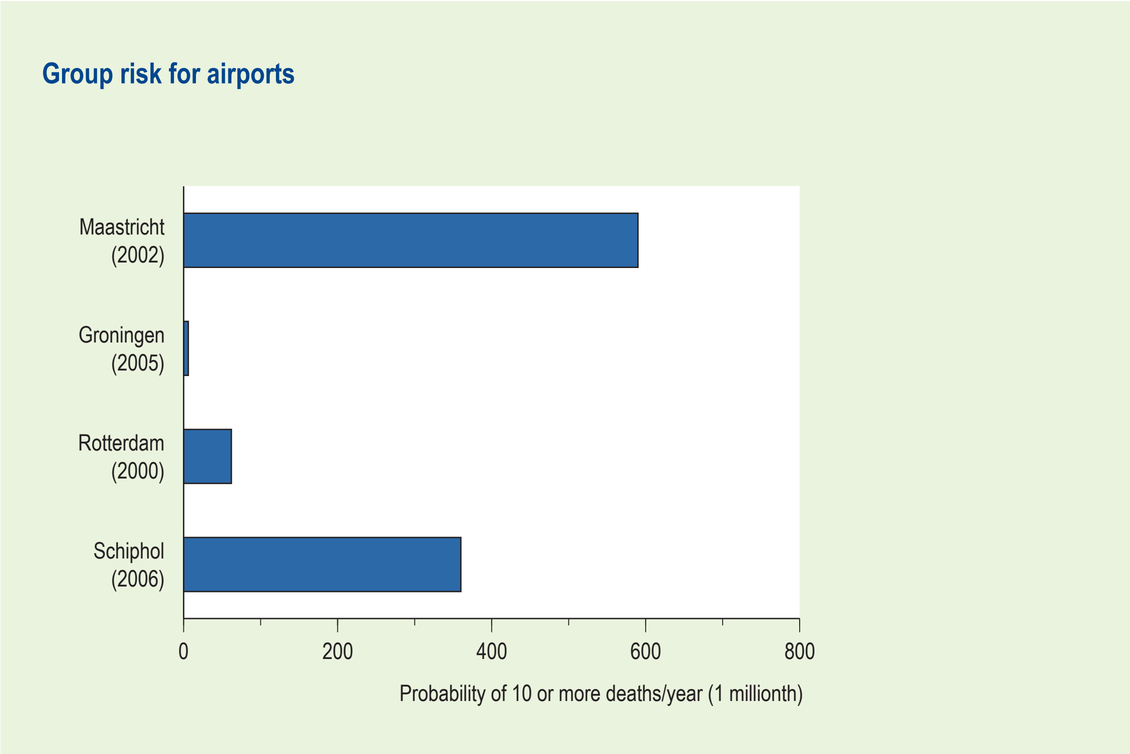

Figure 2.12 Probability of 10 or more deaths resulting from an accident at an airport.

Figure 2.13 Development of location-related risk (left) and group risk (probability of a disaster with more than 40 victims) (right) from air traffic around Schiphol. Calculations for all years are based on actual volumes of air traffic.

External safety

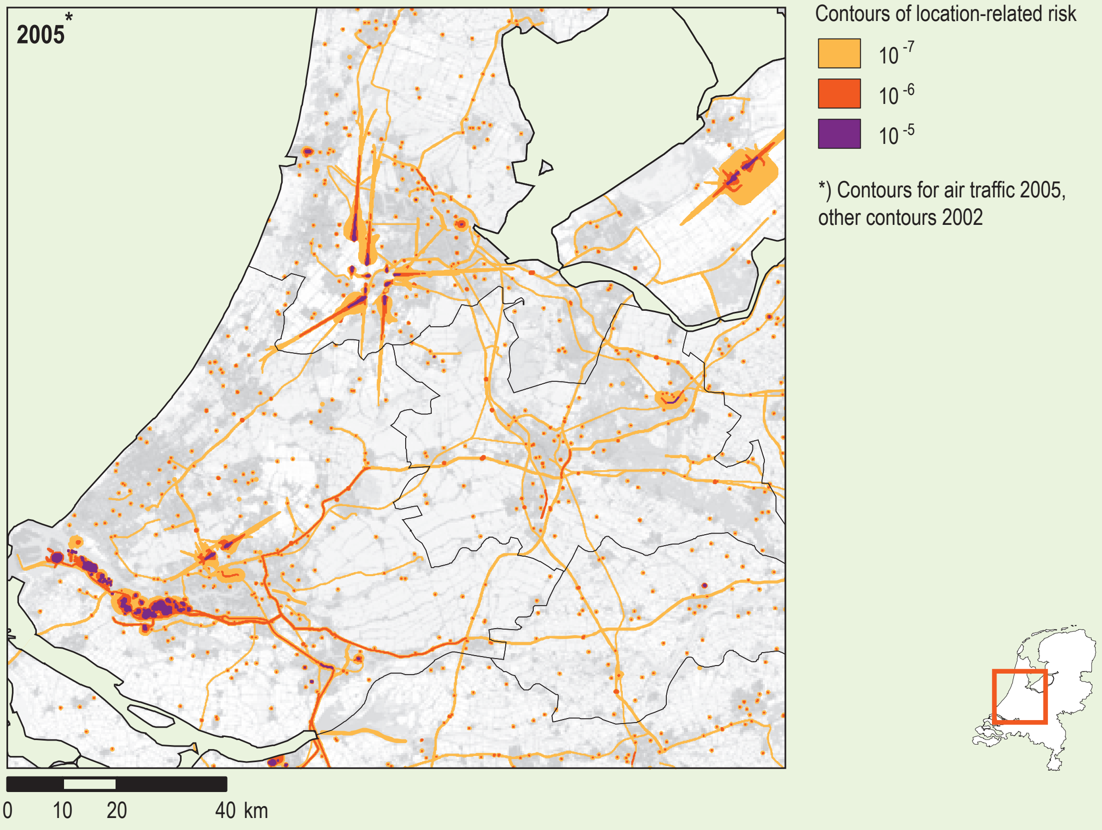

Figure 2.14 External safety risks in the Randstad: contours for roads, rail transport, aviation, LPG stations and safety risk companies.

the Dutch abbreviation EV). External safety risks are determined by the scale and location of a high-risk company/object. For example, chemical companies in large industrial zones contribute far less to disaster risk than the transport of hazardous substances through urban areas or LPG filling stations in residential areas. Figure 2.14 gives a picture of the external safety risks in the Randstad.

Target for dealing with problem areas within the 10^{-5} risk contour around LPG filling stations and safety risk companies

On the basis of current information, the 2007 target has been met for dealing with problem areas within the 10^{-5}/year location-related risk contour around LPG stations and companies required to submit a safety report (safety-risk companies). It is not yet certain whether the target for 2010 for dealing with problem areas within the 10^{-6} location-related risk contour will be met because plans to introduce transport networks for hazardous substances have not yet been realized. Furthermore, there are still a number of external safety problems relating to pipelines. This is discussed in more detail in the *Environmental Balance* 2007.

It is not yet certain whether the target for 2010 for dealing with problem areas within the 10^{-6} location-related risk contour will be met because plans to introduce transport networks for

Table 2.2 No. of people living within a location-related risk contour greater than 10^{-6} in 2005.

Risk source	Number of people within risk contour
Safety risk companies	410
LPG filling stations	7,700
Railway yards	4
Transport by road	40
Transport by rail	2,500
Transport by water	2
Pipelines	Not yet calculated
Total	**10,656**

hazardous substances have not yet been realized. Furthermore, there are still a number of external safety problems relating to pipelines.

In 2004, the External Safety (Establishments) Decree (BEVI) was extended to cover safety-risk companies (large chemical companies that are required to submit a safety report) and LPG filling stations. This means that situations that do not comply with the decree must be remediated within three years. This objective has been met through at-source measures or through the remediation of vulnerable objects located within the 10^{-5} risk contour. The problem areas at other BEVI installations (e.g. ammonia cooling plants) have not yet been precisely identified. The BEVI has applied to railway yards since 2007. The number of people exposed to the 10^{-6} risk contours for natural gas pipelines is not yet included in Table 2.2.

In 2005, more than 10,000 people (Table 2.2) lived within the 10^{-6} contours around high-risk activities such as transport routes for hazardous substances or large chemical plants. For all these people, the risk of becoming the victim of an accident involving hazardous substances is greater than once in 1 million years. The number of people within the 10^{-6} risk contours for natural gas pipelines is not yet known and is therefore not included in the calculations. Of these 10,000 people, 7,700 live too close to LPG filling stations. The large number of people exposed to risks along rail routes (2,500) will have decreased considerably by 2010 due to the opening of the Betuwe Route.

Location-related risks around Schiphol have fallen since 1990

In 2005, almost 600 people in 246 houses in the vicinity of Schiphol were exposed to a risk greater than once in 1 million years of being killed in an aircraft accident (Table 2.2). For approximately 30 people (in 16 houses), this risk was greater than once in 100,000 years. Now that the fifth runway is in use, the number of people living within the 10^{-6} contour has decreased considerably in relation to 1990. The opening of the fifth runway led to a temporary reduction in the probability of a disaster with multiple deaths due to an aircraft accident because there are fewer flights over built-up areas. However, this group risk is higher than in 1990. The probability of a disaster involving 10 or more deaths is now approximately once in 3,000 years (1 divided by 3.5×10^{-4}).

Table 2.3 External safety risks around Schiphol (Dassen and Diederen, 2006).

	1990	1997	2002	2005
Location-related Risk (houses)				
$>10^{-5}$/yr	4	28	21	16
$>10^{-6}$/yr	764	1,513	1,626	246
$>10^{-7}$/yr	8,242	11,211	15,446	4,445
Group risk (probability)				
N>10	3.0×10^{-4}	5.7×10^{-4}	6.4×10^{-4}	3.5×10^{-4}
N>40	3.8×10^{-5}	8.4×10^{-5}	1.3×10^{-4}	6.1×10^{-5}
N>200	2.9×10^{-7}	1.3×10^{-6}	3.7×10^{-6}	2.3×10^{-6}

2.5 Health impact

2.5.1 Environmental health context

Traditional environmental policies and risk management practices have led to substantial improvements of environmental quality. As a result, many risks to human health and ecosystems have been considerably reduced. However, a number of persistent and complex environmental health risk problems still exist, amongst which are air pollution (PM_{10} /$PM_{2.5}$, ozone), noise, external safety (aviation, transport, and industry), microbiological contamination of water, indoor environment, risks of flooding, and soil contamination. In addition, climate change impacts on health may appear in the near future with increased infections, allergies, heat waves and summer smog. Moreover, new and emerging environmental problems and technological innovations might pose health risks that are not yet fully understood. In general, the increased understanding of health risks provided by scientific studies (often showing effects at lower exposures than previously anticipated) and the frequently voiced concern of citizens regarding risks (sometimes alleged or very small) provide input to a broad discussion in society about health risks and safety problems related to current environmental exposures.

Assessing the impact of environmental factors to human health varies greatly depending on the extent of knowledge about the key variables of concern, such as exposure risk and susceptibility factors, and the certainty and nature of the relationship between the risk and the potential outcome. Probably the most familiar method is that of human risk assessment.

The assessment of health effects resulting from exposure to factors in the environment and surroundings also depends on how health is defined. The World Health Organization (WHO) defines health as a state of complete bodily, mental and social well-being, and not just the absence of obvious disorders or diseases. In the RIVM public health model, health is described on the basis of various determinants. In addition to endogenous determinants (inherited or acquired) that can determine human susceptibility, health is also based on exogenous determinants, such as the physical environment and surroundings, lifestyle and the social environment.

2.5.2 Environment and health in a policy perspective

A frequently used paradigm for environmental health policy is the following:

- Decisions, regulatory actions, and legislation
  - Improvement and sustainable development of the quality of the environment
  - Compliance with EU standards and WHO environmental quality guidelines

- Risk management analysis
  - Regulatory options and abatement plans
  - Evaluation of public health, social, and economic consequences
  - Health impact and benefit assessments, cost-efficiency, scenario analyses, multicriteria analyses
- Risk communication
  - Data and information exchange
  - Stakeholders' needs and participation
- Health impact, exposure assessment, and quality of life analyses
  - Quantifying health impacts, exposures, and source contributions
  - Indicators to assess the impact – and benefit – of 'number of lives' and 'healthy life years', including morbidity and annoyance
  - Risk perception and acceptance
  - Sustainable quality of life

Environmental health policy has therefore to deal with a multi-factor, multi-effect phenomenon. A primary focus is on compliance with environmental quality limit values and health-based guidelines. A secondary focus is to strive for a high quality and sustainable environment and quality of life and place.

2.5.3 PBL evaluations of policy on environmental health

The PBL supports national and international decision makers by analysing the environmental impact of policies and of trends in society and acts thereby as an interface between science and policy. The PBL addresses a number of the above-mentioned environmental health issues, strives to develop frameworks and tools for impact assessment and policy analysis, and frequently conducts and reports on impact and benefit assessments. These include publications such as the *State of the Environment*, the *Environmental Outlook* and specific policy evaluation reports.

The following sections describe examples from PBL studies on current and future health risks and impact of policy measures and abatement scenarios.

Air pollution assessment – PM_{10} and ozone (I)
The health impacts of the air pollutants PM_{10} and ozone are periodically assessed for the present and near future in the *Environmental Balance* report (*Milieubalans*, published only in Dutch). A general conclusion is that the health impacts of both air pollutants will probably remain the same or decline only slightly in the years to come, and may even rise due to demographic changes (i.e. an ageing population and increased numbers of the elderly).

Air pollution assessment – PM_{10} and ozone (II)
In the study *Prosperity and the Environment* (*Welvaart en Leefomgeving*, CPB et al., 2006 – available only in Dutch) the health impacts of the air pollutants PM_{10} and ozone have been assessed, both for current and future years, under different proposed economic scenario's up to the year 2040. The example given below illustrates the assessment for PM_{10} in the four largest cities in The Netherlands, shown in Figure 15. Again, it can be concluded that the health impacts of PM_{10} do hardly or not decline in the coming years and may even rise partly due to demographic changes, i.e. ageing and the increased numbers of older people.

Table 2.4 Health impact assessment of PM10 and ozone for the years 2000 and 2002, and the prediction for the year 2010.

	2000	2002	2010 (*)
Short-term health effects	29	29	27
(DALYs per million)	(16-45)	(16-44)	(15-40)
Long-term health effects	10,000	10,000	8,000
(DALYs per million)	(2,000-19,000)	(2,000-19,000)	(1,800-15,000)
Number of days with ozone concentrations above 120 µg/m³ as 8-hr max (average the Netherlands)	10	6	not calculated
Short-term health effects	31	32	34
(DALYs per million)	(14-52)	(14-53)	(15-56)

(*)based on recent measurements, the perspectives on particulate matter concentrations have shifted; consequently, the estimates for 2010 have been reduced by 10-15%

Trends in the environmental burden of disease in the Netherlands, 1980 - 2020

Several aspects of the environment, such as exposure to air pollution or noise, can have effects on our health. In order to gain some perspective on the dimensions of this environment-related health effect in the Netherlands, Knol and Staatsen (2005) have calculated Disability Adjusted

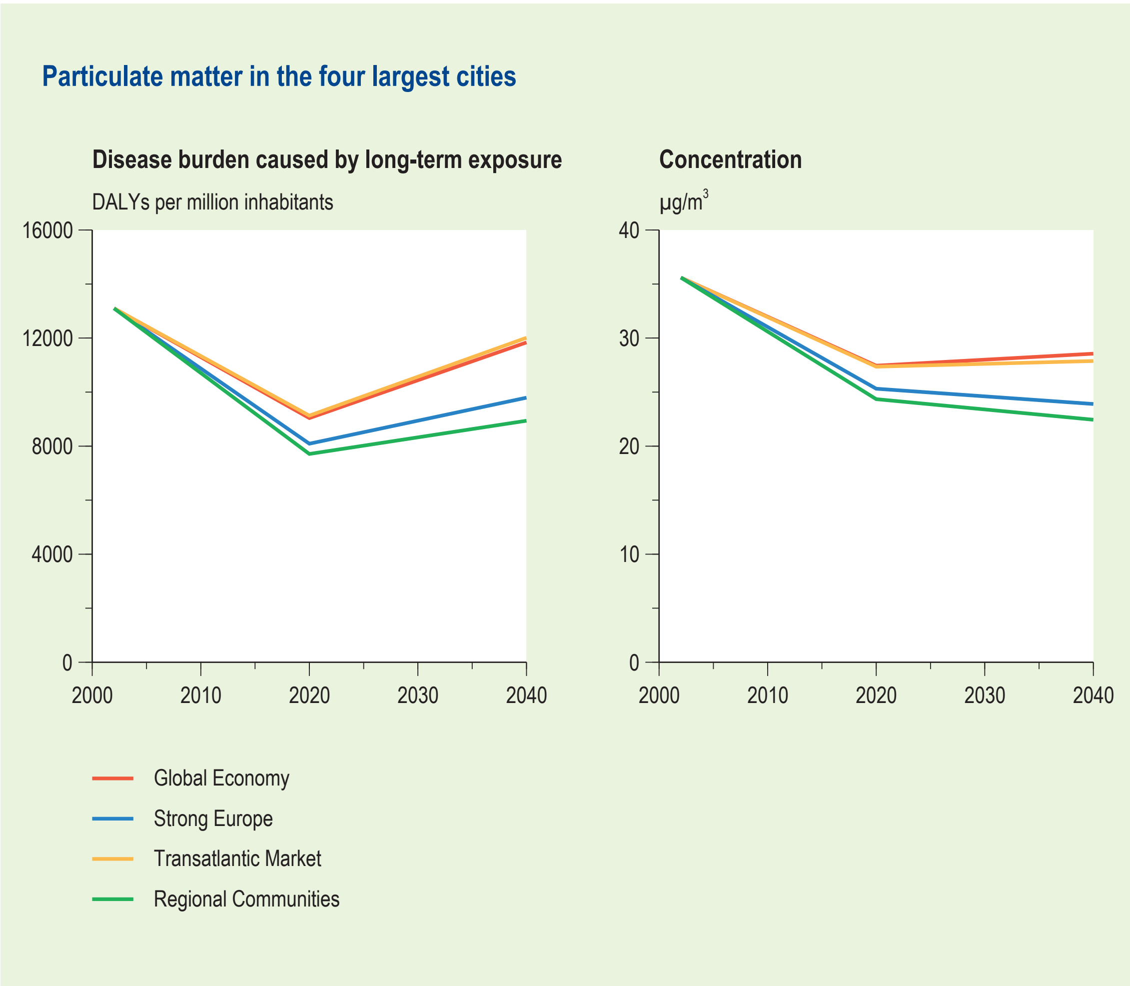

Figure 2.15 Concentrations of particulate matter and health effects of long-term exposure to particulate matter in the four largest Dutch cities (Utrecht, Rotterdam, The Hague and Amsterdam) according to four economic scenarios

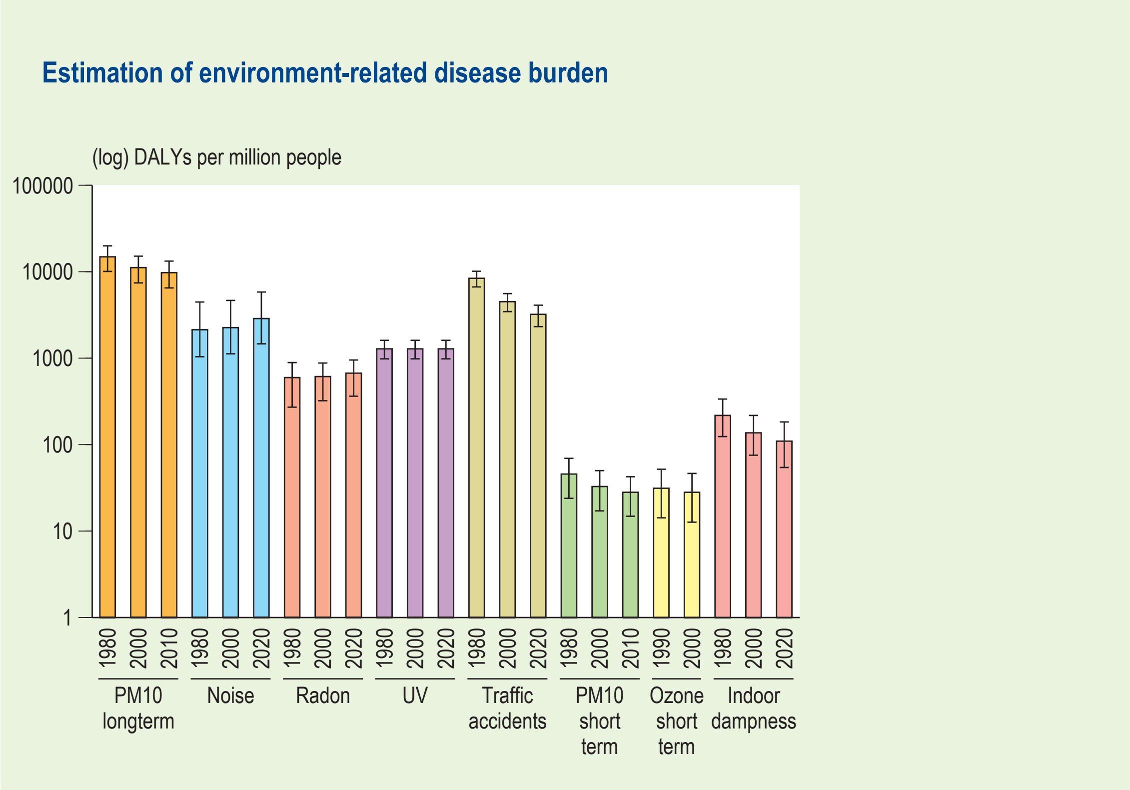

Figure 2.16 Estimation of the environment-related disease burden in DALYs per million people for 1980-2000 (Knol and Staatsen, 2005).

Life Years (DALYs) for the health effects of air pollution, noise, radon, natural UV radiation and indoor dampness for the years 1980, 2000 and 2020. DALYs give a rough approximation of the estimated number of healthy-life-years that are lost in a population due to premature mortality or morbidity (the disease burden). The data are shown inFigure 2.16.

In the Netherlands, roughly 2% to 5% of the disease burden (as calculated for 49 diseases or disease groups) can be attributed to the effects of short-term exposure to air pollution, noise, radon, total natural UV and dampness in houses for the year 2000. Including the more uncertain long-term effects of PM_{10} exposure, this can increase to slightly over 10%, assuming a reference value of 0 μg/m³. A reference level of 20 μg/m³ leads to an estimate of roughly 3% to 9%.

Among the investigated factors, the relatively uncertain effects of long-term PM_{10} exposure have the greatest impact. Long-term PM_{10} is an indicator of a complex urban air pollution mixture. The levels of PM_{10} are decreasing; therefore the related disease burden is also expected to decrease. Noise exposure and its associated disease burden will probably increase up to a level where the disease burden is similar to that attributable to traffic accidents. These rough estimates do not provide a complete and unambiguous picture of the environmental disease burden; data are uncertain, not all environmental-health relationships are known, not all environmental factors are included, nor was it possible to assess all potential health effects.

Consequences for the Netherlands of the EU thematic strategy on air pollution

Particulate matter (PM) is a complex, heterogeneous mixture, the composition of which (particle size distribution, chemical characteristics) changes over time, and is dependent on emissions from various sources, atmospheric chemistry and weather conditions. Currently, limited

Table 2.5 Concentration contribution (in µg/m³) of the 'causal' fraction in PM_{10} and $PM_{2.5}$ in 2020 averaged over the Netherlands and the difference in health impact (HI) by applying Maximum Technical Feasible Reduction (MTFR) when the given fraction is responsible for all health effects.

	CAFE	MTFR	HI
PM_{10}			
Hyp. 1 Total PM_{10}	27.6	21.3	-23%
Hyp. 2 Anthropogenic PM_{10}	9.7	6.3	-35%
Hyp. 3 Primary anthropogenic PM_{10}	4.2	2.7	-34%
Hyp. 4 Primary combustion PM_{10}	1.5	1.1	-29%
$PM_{2.5}$			
Hyp. 1B Total $PM_{2.5}$ [a]	17.9	14.6	-18%
Hyp. 2B Anthropogenic $PM_{2.5}$	9.2	4.6	-50%
Hyp. 3B Primary anthropogenic $PM_{2.5}$	3.5	1.5	-58%
Hyp. 4B Primary combustion $PM_{2.5}$	1.2	0.7	-41%
Ultra-fines $PM_{0.1}$			
Hyp. 5 Estimated ultra-fine	0.7-0.8	0.4-0.6	-20-40%

[a] Assuming a contribution of 10 µg/m³ from unknown and natural sources.

knowledge on the health relevance of the various particle components does not allow a precise quantification on the contribution of different sources and different PM components to the effects on health caused by exposure to ambient PM. Therefore, the current risk assessment practices consider particles of different sizes – from different sources and with different compositions – as equally hazardous to health, and follow the WHO Air Quality Guidelines recommendation to use $PM_{2.5}$ (or PM_{10}) mass concentration as the indicator of health risk. It is expected that better understanding of the potential differences in hazards of particles from various sources will facilitate targeted abatement policies and more effective control measures for limiting the burden of disease.

To support the Clean Air for Europe Strategy of the European Commission, the PBL (Folkert et al., 2005) assessed the consequences for the Netherlands of this strategy on air pollution. For this study, the contributions of different sources to the PM concentrations have been estimated for 2020, on the one hand under the assumption of complete implementation of the CAFE strategy, on the other hand assuming implementation of maximum technical feasible reduction (MTFR). The difference in gain of health has been estimated using five hypothesis on the causal fraction of PM, assuming that the given fraction is responsible for all health effects.

Table 2.5 presents the assumed causal fractions per hypothesis, the concentration contributions of those fractions in 2020 under CAFE and under MTFR, and the difference in health impact.

The analysis of these five hypotheses reveals that quite different abatement policies have to be considered for each of them. However, the control of primary particles from combustion sources and especially from the transport sector seems to be an effective measure under all five hypotheses. This conclusion is also in line with that of the WHO in that particles from combustion sources seem to be particularly important for health risks.

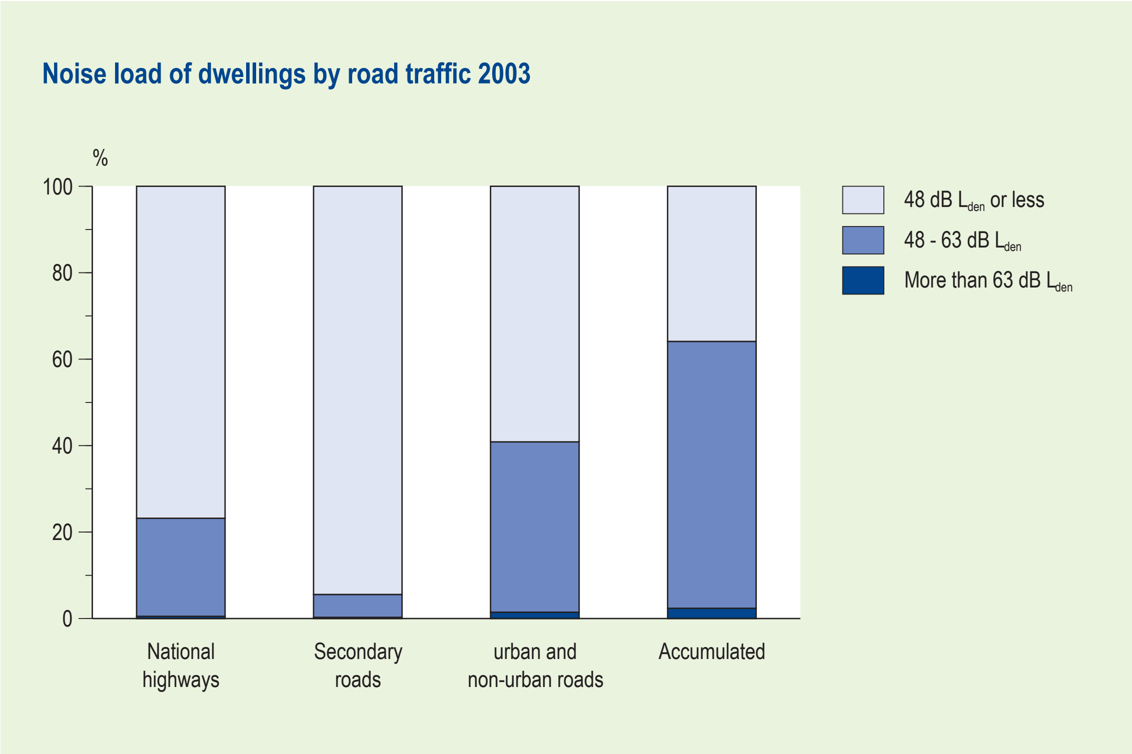

Figure 2.17 Noise load of dwellings caused by road traffic in the Netherlands, 2003 (PBL, 2006)

Noise exposure assessment – traffic
Figure 2.17 shows the noise load of dwellings caused by road traffic. Information about all road traffic in the Netherlands is inventoried or, if not available, assessed. Contributions to the noise level on a grid are calculated with the model EMPARA with regard to the type of road. Combinations of maps with noise levels and a map with the spatial distribution of houses leads to information about the noise load of houses. It is assumed that the result for a single house near a specific road is uncertain. However, results presented in a statistically correct fashion are more accurate and provide a good picture of the situation, which is useful for policymaking.

Noise exposure assessment – Schiphol airport
With 403 thousand flight movements in 2004, Schiphol Airport is the 4th largest airport in Europe. Located between Amsterdam and other cities, noise annoyance is a point of interest (Figure 2.18). Detailed information about flights, including class of aircraft, time of day and flight path, are used to calculate noise levels. Combination of maps with noise levels and houses gives information on the noise loads of houses in a large area around the airport. By using of exposure-response relationships, an assessment of the effects can be made.

***Environmental Balance 2008*: Focus policy on noise level reduction**
Policy that aims to reduce noise levels will lead to a smaller group of people who are exposed to high noise levels, and thereby to a reduction of serious health effects for which a relationship with high noise levels has been established. Due to this policy, the current noise levels as a whole will also decline; and as a result, there will be a smaller group of people in the uncertainty range of effective noise levels for the occurrence of serious health effects. The absolute number of individuals experiencing less serious symptoms, such as annoyance and sleep disturbance, is larger with lower [effective] noise levels. Policy that pursues lower noise levels therefore has a

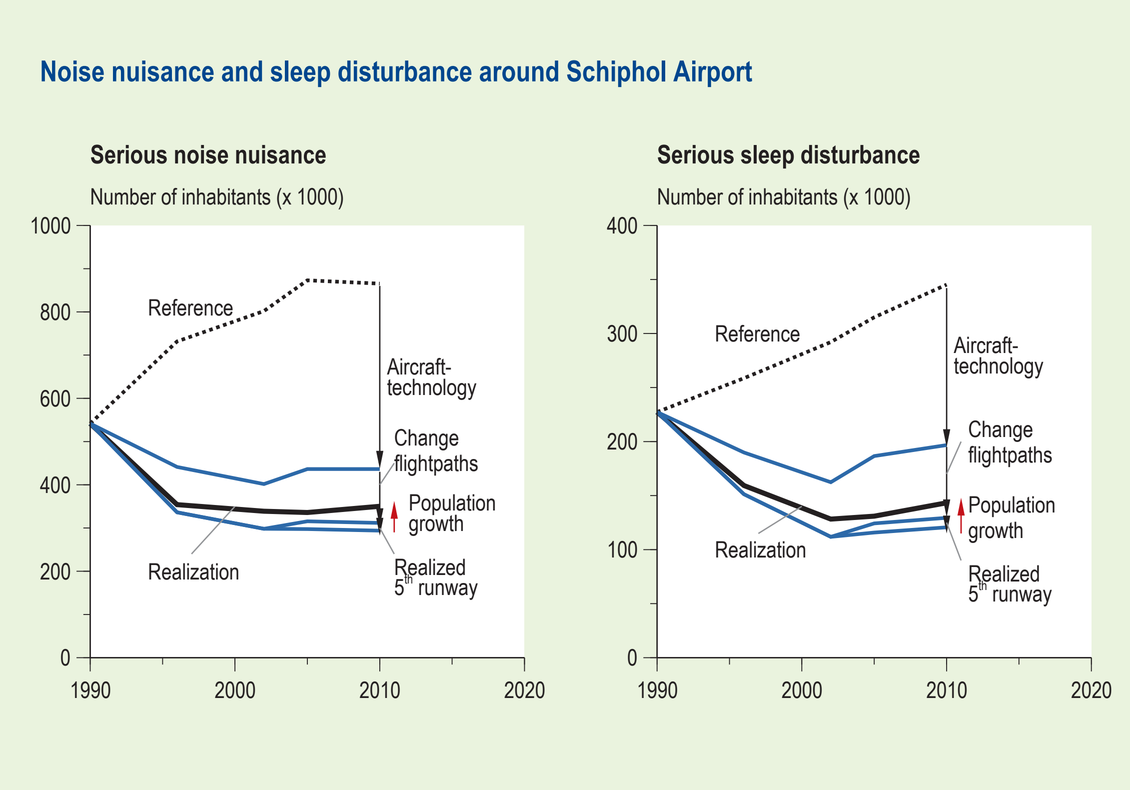

Figure 2.18 Noise annoyance and sleep disturbance around Schiphol Airport: historical and expected realization and effects of relevant developments (PBL, 2005).

larger reach in terms of health benefits. For reducing noise levels, source measures are available such as noise barriers and quieter road surfaces.

Conclusion

Exposure to high noise levels leads to more cardiovascular disease, but it is still uncertain exactly which effects occur at which effective noise levels. Policy that focuses on general reduction of noise levels is effective for reducing serious health effects.

Dealing sensibly with risks – the new concept

Modern, targeted environmental regulation should be based on outcomes and risks (UK Environment Agency), and policies and strategies should maximize the net benefits to society while maintaining fairness and environmental justice for people (NERAM IV Colloquium Statement). Statements such as these add to the ongoing development of a new, multi-dimensional risk analysis approach as a basis for policy evaluation and decision-making. Recently, the Dutch Cabinet has agreed to broaden this relatively new risk policy concept to including urgent health issues at other ministries and to use it as a primary risk analysis and risk management concept. The PBL has agreed to continue the development of various parts of this risk evaluation concept, such as analyzing and integrating psychological and environmental concepts, including risk perception and risk weighing characteristics and preferences, multi-criteria and comparative risk assessment methodologies, and cost-effectiveness and cost-benefit analyses with various restriction features. These restriction features include everyone below the standard, some people but not all below the standard, maximizing health benefits at minimal costs and 'efficiency versus equity' considerations. Examples that will be analyzed using these approaches are health risks from air pollution, aviation, noise, soil contamination and remediation, external safety and flooding. The first analyses in the extended study are expected to be finalized in the third quarter of 2008.

2.5.4 Developing an environmental health planner

Within the PBL, traditional environmental health issues are being expanded to include more sustainable development and quality of life and place concepts, urban liveability matters, and integration of assessments of ecological, economical and social values.

Supported by the EU project INTARESE, the PBL is developing the *Environmental Health Planner (EHP)*, a multi-pollutant, multi-exposure and health impact assessment tool to be used in PBL policy evaluations on environmental quality. The innovative aspect of this EHP is reflected in the description of its functional role: it will be used for ex-ante and ex-post policy evaluations and connect health and wider impacts to causative environmental factors. Development of this EHP tool requires the improvement of quantitative exposure and risk assessment methodologies and therequired data bases, extension of the set of risk metrics regarding mortality, disease burden, healthy or lost life years, quality of life, and risk perception, and finally its use in calculations for abatement scenarios and specific intervention studies, including cost-benefit data on urban, national, and European scales. The further development of this tool and its application in environmental quality analysis is a key part of the PBL programme.

3 Methods and data

This chapter describes the methods and data that are used by the LOK team for the policy evaluation. The dossiers differ in this regard. For air quality and noise, the team uses its own models. For air traffic and external safety, the model calculations are conducted elsewhere and LOK uses only the results for the policy evaluation. Regarding health effects, the results are obtained elsewhere, but the LOK is developing its 'own' model.

3.1 Air Traffic

This section deals with the calculation and analysis methods used in the policy evaluation research. These spatial delineations are the basis for estimating the health effects and risks for residents near Schiphol Airport. A description of the health risk assessments is given in Section 3.5.

3.1.1 General characteristics of calculation models for air traffic noise and external safety risks

To be able to answer the research questions formulated in Chapter 2, Section 2.1.2, the relationships between two sets of data must be understood: 1) the fleet and the fleet composition, flight patterns and runway use and the location of houses and 2) the exposure to noise and external safety risks for local residents and the possible effects on their well-being and health. For its quantitative spatial analysis of these relationships, the PBL uses data from specialized external institutions that collect data and manage and maintain specific models for such analyses. The PBL commissions these institutions to perform calculations and deliver the results. Noise models that comply with the Dutch calculation regulations for aircraft noise are used (Van der Wal et al., 2001). These models have a legal status in the Netherlands. External safety risk models that comply with regulations for a calculation method which is the standard in the Netherlands are used (Pikaar et al., 2000). Both of these calculation regulations were formulated by the National Aerospace Laboratory (NLR). This institute maintains and manages models that comply with the calculation regulations. The NLR does this for the Ministry of Transport and Public Works and the Ministry of Defence.

This section briefly deals with how the spatial delineation of the aircraft noise and the external safety risks of the air traffic are calculated. To a certain degree the models for noise and external safety are similar. Both models determine relationships between the fleet and fleet composition, the environmental and operational performance of the aircraft, flight patterns and runway use. Both models provide a spatial distribution of the noise and the external risks of the air traffic around an airport to a reasonably high degree of detail. The noise model uses internationally available data from aircraft builders about acoustics and flight performance. Using the external risk model, the NLR determines the risk of accidents at various stages of take-off and landing, based on plane crash casuistics (Pikaar et al., 2000). Figure 3.1 shows a schematic diagram of the calculation models for noise and for external safety.

The main advantage of using external models that are in compliance with legal regulations or standard calculation methods is their uniformity and consistency. In the Netherlands, all

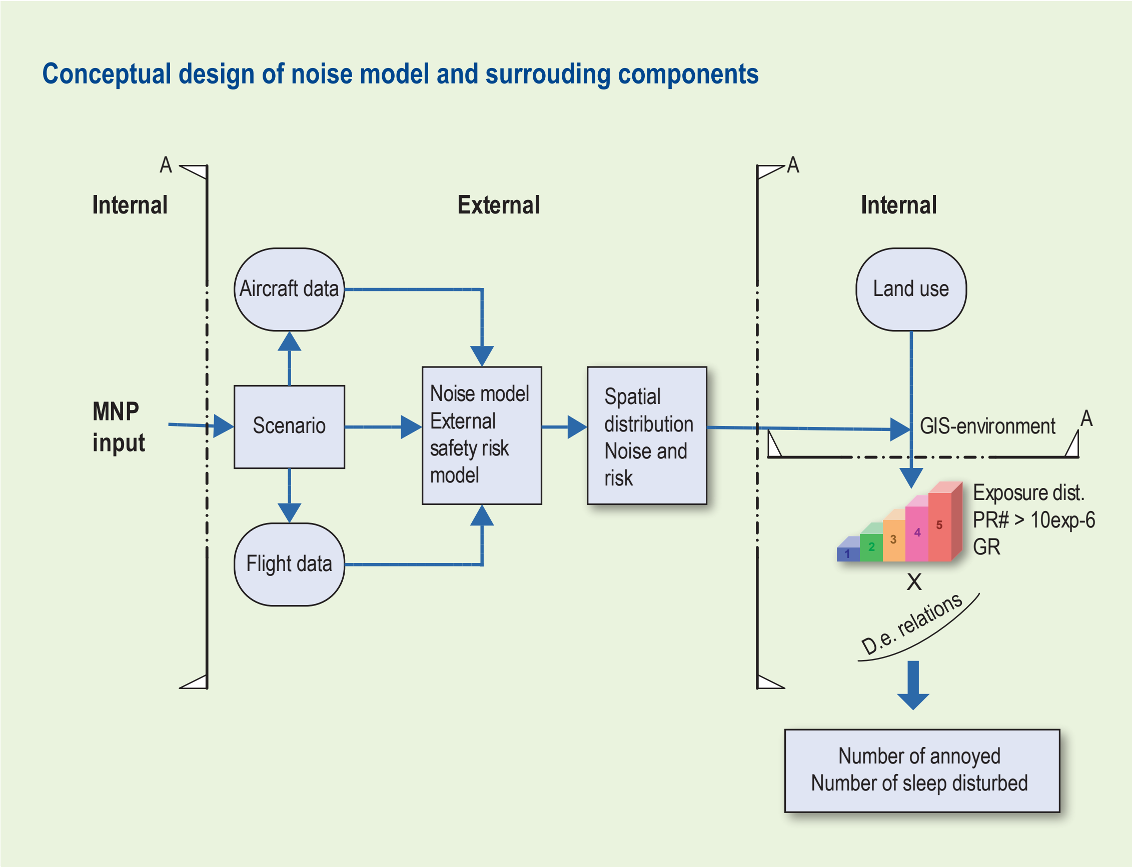

Figure 3.1 Conceptual design for calculation of annoyance and sleep disturbance caused by air traffic.

institutions use certified models – or the results from such models – that are in compliance with the calculation regulations. This applies not only to institutions that perform research into the effects of air traffic, but also to those that are involved in policymaking, policy implementation and law enforcement. These models have also been compared to each other (Peutz, 1997; Resource Analysis, 2000 and BB&C, 2004). This research came to the conclusion that certified models that comply with the Dutch calculation regulations differ little from each other and yield nearly identical results.

The calculation models for aircraft noise and for air traffic external safety risks are an extreme schematization of physical reality. In other words, the fact that computer models generate the same results does not mean that they are 'accurate' in the sense that they present a correct representation of reality in all cases. It should be noted that 'accuracy' as used here cannot be viewed separately from the uniform and consistent use of certified models by all policy bodies and research institutes. For instance, this applies to the research within the framework of the Public Health Assessment of Schiphol Airport (*Gezondheidskundige Evaluatie Schiphol* – GES) that began in the early 1990s. The health effect and risk evaluation methods that the GES research yielded are also used by the PBL (Breugelmans et al., 2005). The consistent use of one model, not only for research but also for making and evaluating policy, ensures that any model inaccuracy will play a subordinate role. This is because a 'balancing' effect occurs regarding systematic errors and inaccuracies which applies to enforcement as well as to health effect evaluation methods.

3.1.2 Reliability of the noise model

Research has been carried out into the degree to which the noise calculation model yields a reliable representation of reality, assuming that this reality can be represented through noise measurements. All research into the differences between calculations and measurements leads to the idea that models calculate lower values for aircraft noise than those determined through measurements (OMEGAM, 1992 and later years; Jonkhart, 1997; Galis, 2000; Eisses et al., 2004; Eisses et al., 2006; Jabben, 2007).

For instance, the Dutch model for aircraft noise does not take into account the direction-dependent noise radiation from an aircraft, the meteorological influences (wind and temperature) on the propagation of sound over longer distances, and the possible influence of buildings on the propagation of sound. The effects of the ground (reflection, diffusion and attenuation) on the propagation of sound is taken into account in a simplified way. In addition, standard manufacturing data on the flight performance of a limited number of aircraft (take-off and landing speeds) and their noise production are used. For aircraft for which no data are available, it is assumed that the performance and noise characteristics are identical to those of aircraft for which such data are available. Finally, it is important that such data are determined for relatively small distances between the aircraft and the recipient (to a maximum of a few kilometres, depending on the type of aircraft). The calculation points within the noise study area mostly involve a longer distance to the flight path, which means the performance and noise data must be extrapolated.

At the beginning of the present decade, the debate about the reliability of the calculation model intensified. The PBL therefore decided in 2002 to have the Netherlands Organization for Applied Scientific Research (TNO) conduct a comparison between calculated and measured values for aircraft noise near Schiphol Airport. This entailed measuring a reasonably large number of aircraft taking off and landing. Based on the aircraft recordings and the radar tracks, the corresponding maximum noise levels (LA_{max}) and noise exposure levels (LAX) were calculated. This study confirmed the already existing idea that, on average, measured results are higher than calculation model results. The average differences between the measured and calculated maximum noise level LA_{max} of aircraft passages are 3 to 4.5 dB(A) for different microphone positions. The differences are smaller for the noise exposure level LAX (used to derive the noise levels L_{den} and L_{night}): between 0 and 2 dB(A).

In 2000, the government appointed a committee (CDV) that was assigned to look into the use of measurements for establishing the norm and for enforcement. This committee conducted the most complete and consistent study into the reliability of the calculation model to date. The study used the results from automated measuring systems for an entire year and from controlled, supervised measurements carried out on a limited number of measuring days. The biggest differences were found between the calculation results and results from NOMOS, the automatic measuring system at Schiphol Airport. The study shows that the difference between the calculated and the actual aircraft noise level is relatively small at some locations, but that it can be as much as 5 to 10 dB(A) at others. The average difference is between 3 and 4 dB(A). The average difference between the daytime noise level LA_{max} of aircraft passages and the corresponding calculated levels can be more than 10 dB(A). The measured values for the noise exposure levels LAX (which is used to derive the noise level L_{den}) are also frequently higher than the calculated values. However, the differences for LAX are smaller than those for LA_{max}.

The CDV study did not exclude the possibility that disturbance played a role. However, at three of the five locations where measurements were being carried out under constant supervision (and where a significant influence from distortion can be excluded), large differences between measured results and calculated results were also found. Therefore, the CDV concluded that there are clearly other factors playing a role in the differences between measured and calculated aircraft noise. The final report discussed a number of possible shortcomings of the calculation model, including it being an extreme schematization, and the use of manufacturer's data about aircraft flight performance and noise production.

The CDV made an interesting comment about the ascertained differences between measurements and calculations. "With regard to transparency and clarity to local residents, this [the differences between measurements and calculations] is undesirable. The problem is not solved simply by changing to a different calculation model (such as INM). Improvements to the current calculation model will certainly contribute to the solution. This mainly concerns the description of the noise source (the aircraft) with the actual flight procedures and the description of the propagation of sound under specific conditions."

As far as is known, there has been only one study into the differences between the Dutch model and alternative models used in other countries (Van Kessel, 2004). This is a study from 2004, which concerned the differences between the National Model (NM), the Integrated Noise Model (INM) and the European model (in other words, the ECAC model) (ECAC.CEAC, 1997).

The study showed that ECAC and INM differ little with respect to design and physical schematization. The study made the following conclusion: "ECAC and INM differ in detail as regards the points from the aircraft categories, the degree to which atmospheric conditions and surrounding features are taken into account, and how contours are determined. In many ways, the ECAC is a simplified version of INM. The differences between these models and the NM are much greater. They mainly pertain to the degree to which different type of aircraft are assumed to fly and produce noise in the same way. The NM assumes many types of aircraft are the same. In addition, there is a significant difference with regard to the distribution of the routes flown in segments. The INM and ECAC use a segmentation based on flight routes, while the Dutch model uses one that is based on segments of equal duration and procedures. There are also differences in the model for soil attenuation and in the model for the effects of wind, temperature and humidity on the propagation of the aircraft noise to the ground. The NM uses even fewer corrections for this than the ECAC and INM.

The quantitative comparison of the Dutch model and INM shows that, for the compared situations, the NM calculates a higher value (2.6 ± 1.0 [dB(A)]) for the noise level than INM for many types of aircraft. This finding applies to a specific situation in which a few dozen landings were simulated and the noise level was calculated at four points, the shortest distance to the flight path being a few hectometres. Given the encountered difference, it is advisable to investigate the difference between INM and NM for several combinations of location/type of aircraft/flight. This investigation should map out the differences that can be expected for the entire area around Schiphol Airport if the NM was 'traded in' for INM."

It should be noted that the comparative calculations were done using INM version 6.0. There is now an updated version of INM available (version 7.0).

Table 3.1 Sensitivity of local risk (area inside contour and average value) for a number of model parameters.

Model parameters	Location-related risk area in relation to nominal value	Location-related risk average value in relation to nominal value
Risk of accident	40 – 200%	20 – 300%
Changes in the distribution functions or their parameters	75 – 115%	85 – 140%
Accident spread in relation to the flight path	115 – 125%[1]	70 – 85%[1]
Size of crash area	85 – 115%	80 – 120%
Combination of parameters	30 – 205%	80 – 120%

[1] for location-related risk 10^{-6} contour

3.1.3 The reliability of the model for external safety risks

The external safety model has also been reviewed in the past (Pikaar et al., 2000; BB&C, 2004). An important point from these reviews is that in 'standard' calculations with the model IMU [1)], the nominal (i.e., the 'average') values are used for all parameters. However, it appears that the reliability is mainly influenced by the relatively small number of accidents representative for Schiphol Airport, per type of accident scenario. The limited accident casuistics means that a number of aspects can only be determined with relatively large uncertainty margins. These aspects include the accident risk, the distribution functions and their parameter values, the spread of the accidents in relation to the flight path, the type and extent of the crash area and the death rate within the crash area.

The analysis by BB&C (2004) mapped out the sensitivity to uncertainty in the various model parameters by calculating 5% and 95% confidence intervals for the parameters. This analysis looked at the sensitivity in the local risk contours, both in the contour area and the average location-related risk value in a certain region. The study showed that sensitivity is highest with the accident risk parameter. Other parameters yield smaller changes.

Table 3.1 lists the study results (from PBL(a), 2005). Apparently, the contour area at a minimum accident risk (5% confidence interval) is slightly under 40% of the area at the nominal risk. At a maximum accident risk (95% confidence interval), the area is almost 200% larger than with a calculation at the nominal risk. For the average location-related risk value, the change is even larger: 20% at minimum risk and 300% at maximum risk. The last row in the table lists the impact of a combination of parameters, such as the accident risk, the distribution and the crash area. This provides a more complete picture of the sensitivity of the model than single parameter variations. The statistical variance in the accident risk is dominant here.

Exposure and health risks

In order to estimate the exposure, the residential and work locations of residents around Schiphol Airport have been meticulously mapped out for the years 1990, 2002, 2005 and 2010. Appendix 5 (*Modellering van het ruimtegebruik*) in MNP(a) (2005) describes in detail how this was done. Due to its high degree of accuracy, this modelling method was adopted by the

1) Since 1993 the model IMU is in use and has been adapted a number of times. The model calculates the location-related risk (PR) on the basis of plane characterisations (accident chances by plane generation, maximum take off weight (MTOW), accident impact), use of runway and flight paths. In combination with the population data also group risk (GR) is calculated.

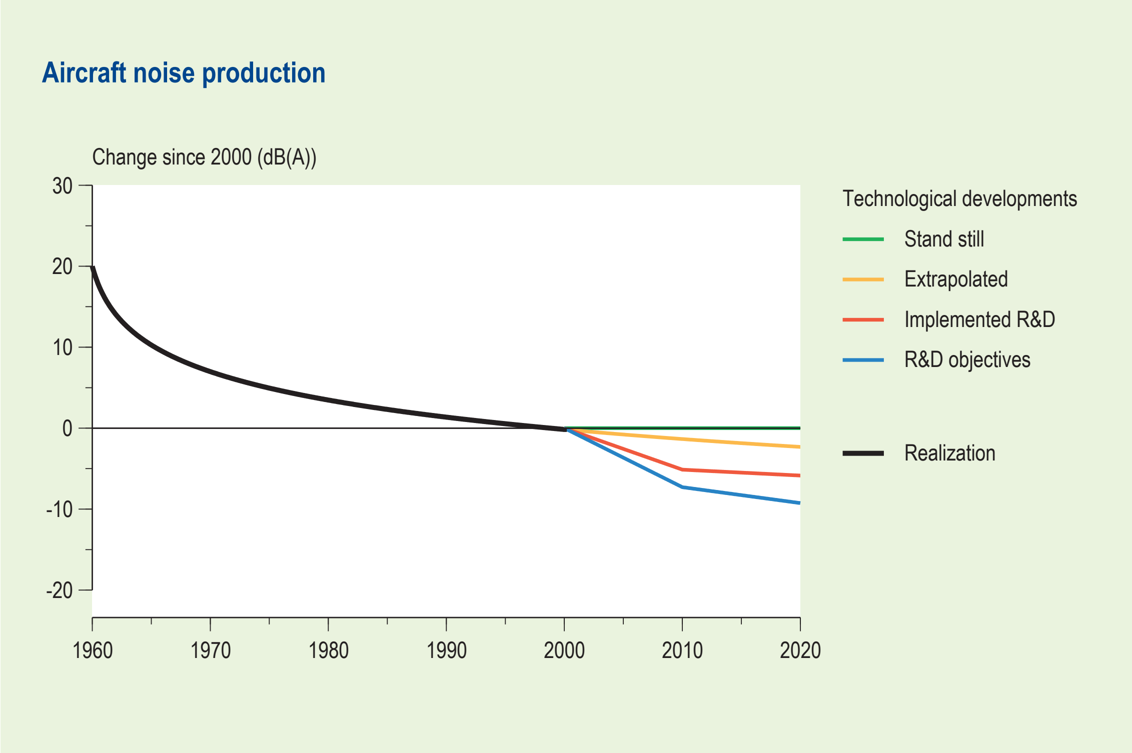

Figure 3.2 Expectations for the noise production of aircraft marketed after 2010.

Ministry of V&W (Transport and Public Works) and used to modify the environmental norms (Vinkx, 2007). The exposure is determined by confronting this modelling method with the spatial distribution of the noise and the external safety risks.

Appendix 2 of MNP(a) (2005) describes in detail how the health effects resulting from exposure to aircraft noise (annoyance and sleep disturbance) are determined and which uncertainties play a role in this process. RIVM researchers involved in the health study around Schiphol Airport contributed to this Appendix. The analysis of the evaluation methods for the extent of serious noise annoyance and serious sleep disturbance showed that the uncertainty in the dose-response relationships for these effects determines the uncertainty in the magnitude of the effects. The 5% and 95% confidence intervals result in estimates of the number of people suffering serious noise annoyance and sleep disturbance that deviate 10% to 20% from the nominal value. Such deviations can be recognized in the reported numbers of people suffering from serious noise annoyance and sleep disturbance.

3.1.4 Prognosis

Analyses of future developments are very sensitive to assumption about the future noise production of the fleet and the future risk of accident. Therefore, the PBL arranged for a number of international air traffic institutes, aircraft and aircraft engine builders to be contacted about their estimates (Wubben et al., 2005). This study led to a much broader bandwidth in the expectations about future aircraft noise production (see Figure 3.2).

The various expectations were then processed into different scenarios for the development of the fleet. The is compatible with the usual methods for scenario analysis.

A comparable study was done for external risks (Ummels et al., 2005). The study showed that the accident ratio of third generation aircraft worldwide has decreased in the past five years and was on average lower than in the period before that. This means that on the basis of casuistics until 2004, the accident risk for third generation aircraft is expected to be lower than that based on the casuistics until 1998. For the near future (until 2010), it is expected that the accident risk for third generation aircraft will decrease further. However, it is likely that the accident ratio will occasionally be higher than in the previous year/years. For the more distant future, the study does not yield any accurate trend prediction because not enough data are available for the specific types of aircraft on the market since 1997. In general, it is expected that the total accident risk of air traffic worldwide will decrease. This will mainly be due to technological progress and phasing out of older planes.

3.2 Noise

Delineation: noise from road traffic and rail traffic (noise from air traffic and industry are not calculated with EMPARA; the data are provided by third parties).

By using the Noise tool in EMPARA, noise maps are calculated on a national scale for road traffic and rail traffic. The noise maps consist of a number of different layers for different sources: rail traffic, motorways, secondary roads, municipal roads and two layers for non-urban roads. These layers are based on various data sources. Depending on the indicator, the maps are also accumulated, with noise maps of air traffic and industrial noise as well.

3.2.1 Indicators

Standards for noise level
Noise level indicates the average sound pressure during a specific time period with respect to a reference level, expressed on a logarithmic scale: the decibel (dB).

Sound comprises a spectrum of frequencies ranging from low to high. In order to arrive at a single number descriptor for sound, the energy from various frequency bands is added together. During this process, it is customary to apply a weighting so that the level is adapted to the sensitivity of human hearing for sound. In practice, the so-called A-weighting is used. The unit of sound is then expressed as dB(A).

Sound is an instantaneous phenomenon, and there are various ways to describe a continuously variable noise load as a single level. The simplest and most widely used method is to determine an average value during a specific time duration.

To describe environmental noise due to road and rail traffic, the prevailing method is to use a average annual level. Traffic data are therefore based on an annual average. To indicate the effect of factors such as weather conditions on noise transmission, an average annual value is used where possible.

The noise level is also determined using a 24-hour average, where a penalty factor is often added during the evening and night. Various noise standards are used that are the result of different weightings of the distribution of noise during a 24-hour period.

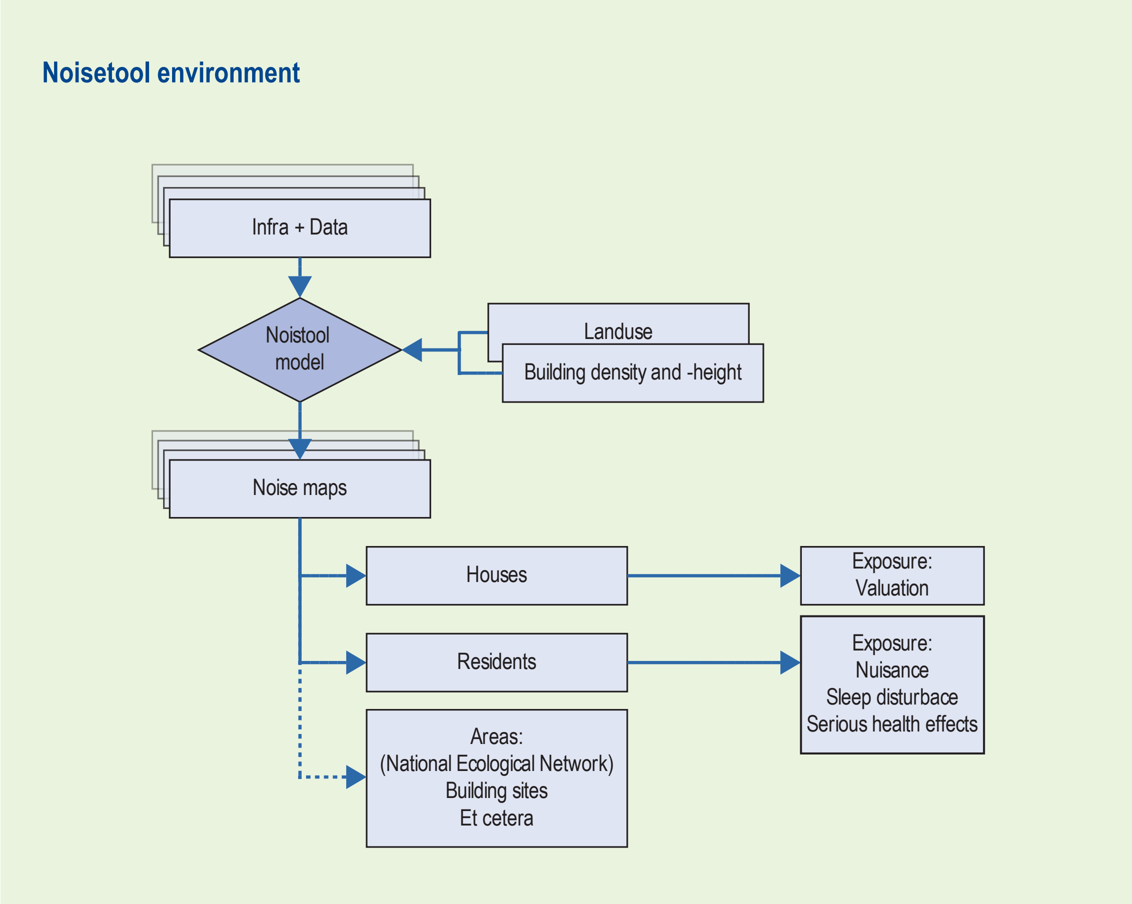

Figure 3.3 Schematic summary of the Noise tool environment.

In the noise maps that are calculated with EMPARA, the noise level is expressed as a generally accepted standard for the sound pressure level: L_{den}. This noise standard is a weighted average for the day, evening and night periods, where a penalty factor of 5 dB is applied during the evening, increasing to 10 dB at night.

Until a few years ago, it was customary in the Netherlands to express noise level for legislative purposes as $L_{etmaal;}$ this is the maximum noise level during the day, evening or night, including the same penalty factors. For policy concerning noise abatement areas, penalty factors are not applied; a 24-hour average level (L_{24h}) is used without penalty factors. As part of research into health effects, sometimes only the noise level during the day or night (L_{night}) is taken into consideration.

For indicators that are based on a different noise standard than L_{den}, the PBL uses generic conversions. The differences between noise standards are determined by the distribution of noise during a 24-hour period. This distribution is different for main roads or railroads than for smaller roads. For conversion to a different noise standard, a characteristic 24-hour distribution of the corresponding source is assumed. In principle, maps in L_{den} are therefore converted to a different noise standard for each map layer (motorways, secondary roads, urban roads, etc.).

Indicators (spatially aggregated noise level)

For the indicators in noise policy, spatially aggregated noise levels are used.

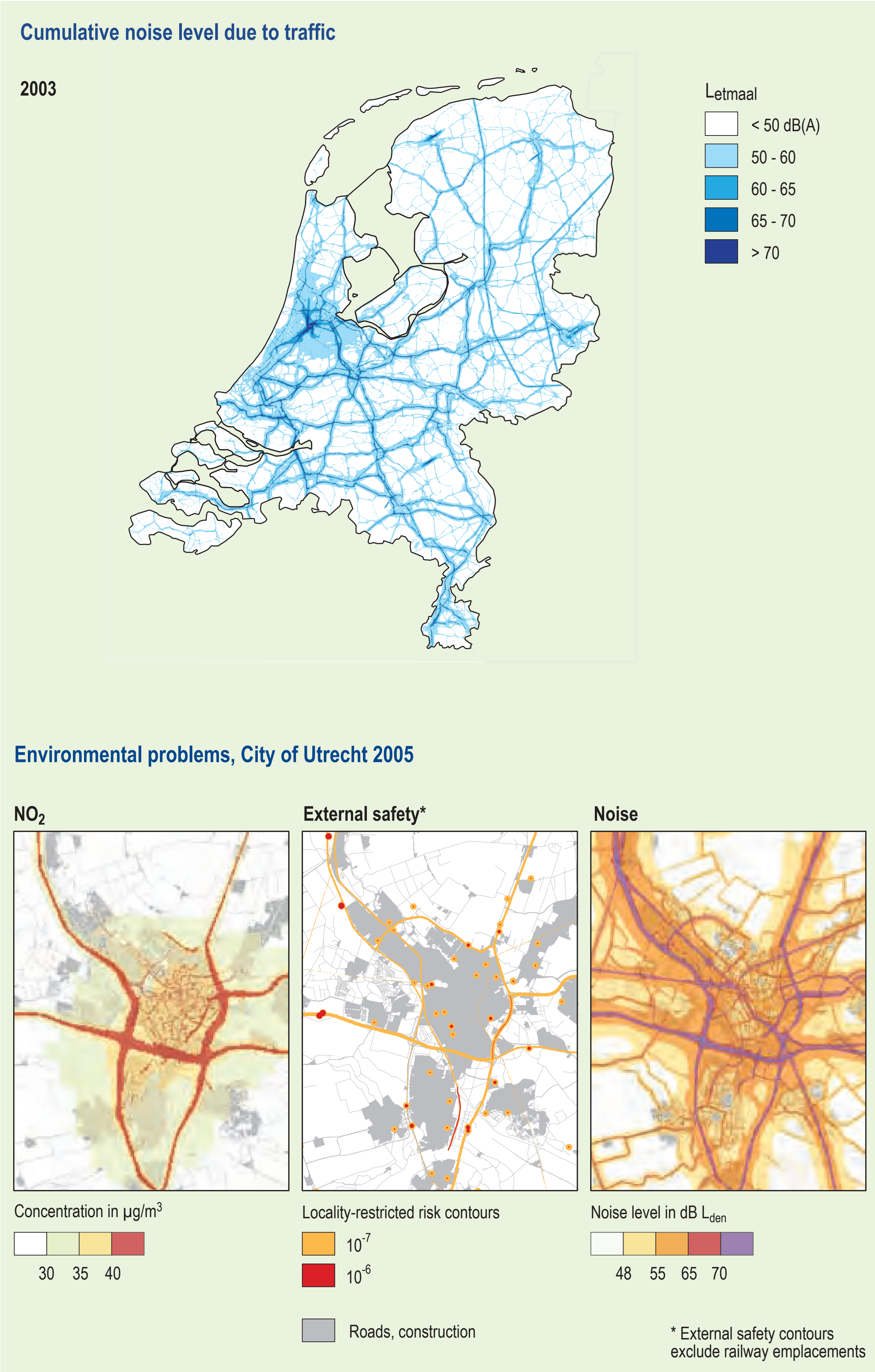

Figure 3.4 Cumulative noise level due to traffic (above), and noise level in relation to air pollution from traffic and external safety near the city of Utrecht. (Source of upper map: RIVM, NLR, AVV, AEA Technology)

In connection with Dutch policy, the PBL often determines the number of houses with a noise level above a specific level. The policy often focuses on “hotspots”: houses with a relatively high noise level (65 dB near roads, 70 dB near railroads). But it also looks at the numbers of houses above a statutorily determined preferred value: (48 dB near roads, 55 dB near railroads.)

Regarding noise abatement areas and nature reserves, the area with noise level above or below a specific value is usually determined. The policy often expresses this noise level in units of L_{24h}, a noise level without penalty factors, and the corresponding sound levels are low.

For policy evaluations, the PBL calculates health effects such as severe annoyance and sleep disturbance, especially as a comparison in various scenarios. The exposure of houses is the point of departure for these calculations.

3.2.2 Model description

The description of the model is largely based on Dassen (2001).

In the Netherlands, there are statutorily prescribed calculation and measurement regulations for acoustic research that takes place in the context of legislation.

These regulations actually consist of two models. Firstly, there is a simplified model (known by the Dutch abbreviation SRM I) that is based on a non-spectral calculation. The use of SRM I is limited to relatively simple situations, for example long, straight roads without noise barriers. For more complicated situations, a second model (SRM II) is prescribed. SRM II calculates using octave bands; this allows it to determine the effect of noise barriers, for example.

The EMPARA model is based on SRM I. For calculations at the national scale, this choice is necessary to keep the data files manageable. The detail level of SRM II is unfeasible for making calculations at the national scale.

With respect to SRM I, EMPARA has been expanded to include noise protection from barriers or other objects like buildings. This type of expansion applied to the SRM I method is often referred to as “Method 1.5”. This means that it does not use spectral modelling, as in SRM II. From source to receiver, sound is expressed in A-weighted decibels. The parameterization of the sound propagation is adapted to the attenuation of standard spectra for road noise and railway noise .

Another reason why EMPARA deviates from the prescribed calculations is that EMPARA can be applied more widely than SRM I. For example, with some indicators it is necessary to determine noise levels at large distances. Important sound sources such as motorways produce a noise level at large distances; this can result in serious annoyance or can exceed the limits for noise abatement areas. In order to map this out successfully, transmission formulas are required which have also been validated for larger distances than the validity area of SRM I. Moreover, it turned out that the noise propagation from road and rail traffic was not included in the calculation regulations with corresponding definitions. For EMPARA modelling, modifications were included based on expert recommendations; this allowed the noise propagation from road and rail traffic to be approached with the same definitions and validated for large distances. This has led to limited modifications of the model for road traffic with respect to the calculation and measurement regulations. These modifications can be found in the modelling of emission and ground attenuation.

The noise level at a location is determined by the emission of sources in the surroundings and the transmission from source to receptor point. This noise level is expressed in terms of decibels; this unit is based on logarithms and results in a simplified formulation.

$$L_{p,den} = E_{den} - T$$

Section 3.2.4 describes the emission (E) from infrastructure in EMPARA. Section 3.2.5 describes the sound transmission (T) in greater detail .

3.2.3 The EMPARA roads database

Road traffic makes a major contribution to local environmental load due to air pollution and noise. In order to successfully map out this contribution, it is important to have access to a sufficiently reliable roads database. The roads database, as described below, is used in EMPARA for both noise and air quality calculations.

For the Netherlands as a whole, a National Roads Database (known by the Dutch abbreviation NWB) is available. This database contains a very precise geometric description of the position of the roads, linked to other data such as the name of the road and the road maintenance authority. The NWB from 2000 is used as the fundamental database for all road data in EMPARA. However, this requires other data to be linked to this database, which are necessary for each road or road section to calculate the emission and dispersal of air pollution and noise. Due to differences in the origin of this data, sub-databases have been established, which differ in the quality of the data. The total roads database is comprised of sub-databases for motorways, secondary roads, non-urban roads and inner-city roads. These sub-databases are briefly described below. There is also a separate discussion of how the traffic intensity calculations are made and how vehicle speeds are dealt with.

The sub-database *main roads* comprises all motorways in the Netherlands. In past years, traffic intensities and the proportions of heavy vehicles have been based on a combination of vehicle counts and calculations. Data about noise barriers are obtained from Directorate for Public Works and Water Management (the road maintenance authority).

The sub-database *provincial (secondary) roads* comprises virtually all roads in the Netherlands that fall under the maintenance authority of the provinces. In general, these are the most important thoroughfares outside the built-up area. Occasionally, secondary roads run through the built-up area.

The sub-databases *non-urban roads I and* II comprise some of the roads outside the built-up area that do not fall under the maintenance authority of the province or national government. Generally speaking, these are roads with lower traffic intensities than the secondary roads. The roads in sub-database I are from regional models[2)] and are linked to the NWB. The roads in sub-database II were added later on. They have a traffic intensity under 5000 vehicles per 24-hour period. These roads were also obtained from the regional models, but are not linked to the NWB.

2) The New Regional Models (NRMs) are traffic models of the regional departments of the Directorate for Public Works and Water Management. The six NRMs differ in quality.

Table 3.2 Calculation methods of traffic intensities for the sub-databases of the EMPARA roads database.

	2006	Prognosis
Motorways	Counts, supplemented with calculations; for 2006 the basis year 2004 was used unchanged.	2010 and 2020 calculated with LMS. 2015 interpolated. New roads are being included in the database.
Secondary roads	Calculations by regional models, calibrated with counts. Basis year 2000, extrapolated to 2006 with indexes for each LMS zone. The proportion of light, medium-heavy and heavy vehicles (specific for each road) is kept constant during this process.	Extrapolation to prognosis years with indexes for each LMS zone. No new roads are being added.
Non-urban roads	The same as for secondary roads	Extrapolation to prognosis years with indexes for each LMS zone. No new roads are being added.
Urban roads	For 2000, generic calculation of intensities based on road class and size of municipality. For large thoroughfares, corrections are implemented with intensities from the regional models. The proportion of light, medium-heavy and heavy vehicles is determined generically for each road class and is held constant during extrapolation to later years. Extrapolation to 2006 with indexes for each LMS zone.	Extrapolation to prognosis years with indexes for each LMS zone. No new roads are being added.

Sub-database II was added to fill in the gaps on the map for noise calculations for the analysis of noise abatement areas.

The sub-database *urban roads* comprises the thoroughfares and local access roads within the built-up area of the urbanized centres in the Netherlands. Traffic intensities are calculated based on the road class [3)] and the size of the municipality. For individual road sections, the calculations can deviate strongly from reality, but on average over a larger area (urban district), the traffic intensity is calculated with reasonable or good accuracy. For the larger thoroughfares, the traffic intensity has been obtained from the regional models. Separate data fields in this database are the road type and the distance to the façade. These are important for calculating the dispersal of air pollution in the streets, see Section 3.3.4. These data were obtained from a combination of the "Top10" vector database of the Topographical Department and the Netherlands Address Coordinates Database of the Dutch Land Registry Office.

Traffic intensities

Table 3.2 provides an overview of the methods with which the traffic intensities are calculated for the sub-databases. An important instrument for determining the development in the traffic performances is the National Model System (abbreviated in Dutch as LMS). This is a computer model that the Transport Research Centre of the Ministry of Transport, Public Works and Water Management uses to make mid-to long-term prognoses (10 - 20 years) for passenger traffic for the Netherlands as a whole. In this model, the country is divided into 345 zones, within which the LMS calculates the traffic performances based on factors such as the economic and demographic developments in the zones. The LMS focuses on making reliable prognoses for motorways. For the underlying road network, the reliability of the prognosis declines rapidly as the network becomes more detailed.

3) The road class indicates the relative importance (and thereby the traffic intensity) of a road. This road classification comprises six levels, ranging from a main road to a local, neighbourhood road. The road class and the corresponding traffic intensity were obtained from the BasNet (Basic Network), a predecessor of the NWB. No reference for this source is still available.

Table 3.3 Summary of the data in the sub-databases of the EMPARA roads database, with an indication of the quality[1]. The indication is based on the data for 2006. For prognoses, the quality of most data is lower.

	motorways	secondary roads	non-urban roads I	non-urban roads II	urban roads
Data					
Intensity	++	0	0	0	0/-
proportion of medium-heavy vehicles	+	0	0	0	-
proportion of heavy vehicles	+	0	0	0	-
24-hour distribution of traffic	+	0	0	0	0
Speed	+	0	0	0	-
distanced to façade					+
Road type					+
road surface type	+	0/-	0/-	0/-	-
Noise barriers	+	0/-	-	-	-
Quality of sub-database					
Completeness	100%	95%	10%	10%	30%
Geometric precision	++	++	++	-	++

[1] quality: ++ very good, + good, 0 moderate, - poor

For secondary roads and the other non-urban roads, the intensity has been calculated in past years using regional models. For these calculations as well, the uncertainty increases as the road network becomes more detailed. The regional models do not provide any prognoses that are usable for relatively detailed noise or air-quality calculations. For all other roads besides the motorways, the intensity for prognosis years is therefore scaled with the developments in the LMS zones.

Moreover, the 24-hour distribution of traffic is important for noise policy. For motorways, this 24-hour distribution is known from traffic counts. For other roads, a fixed distribution during the 24-hour period is assumed, which depends on the road type. As a rule, the percentage of traffic during the night is smaller for roads with less traffic.

Traffic speed
The emissions of noise and air pollution depend on traffic speed and its dynamics. Traffic speed is based on the legal speed limit. Data about the dynamics of traffic speed (such as traffic congestion) are not available in the database. For situations with relatively large dynamics in traffic speed, this can lead to relevant deviations from the actual situation.

Table 3.3 provides an overview of the data in the sub-databases and a global evaluation of the quality. Figure 3.5 shows the sub-databases on the map of the Netherlands.

3.2.4 Emissions

EMPARA uses the emission formulas of SRM I from the calculation and measurement regulations. The definition is: the power of a line source, per metre.

For railroad noise, the calculation regulations determine noise emission based on 10 types of trains. The Dutch railway maintenance authority PRORAIL publishes an annual acoustic report (“acoustical time table”). This report contains calculation programming and databases that are

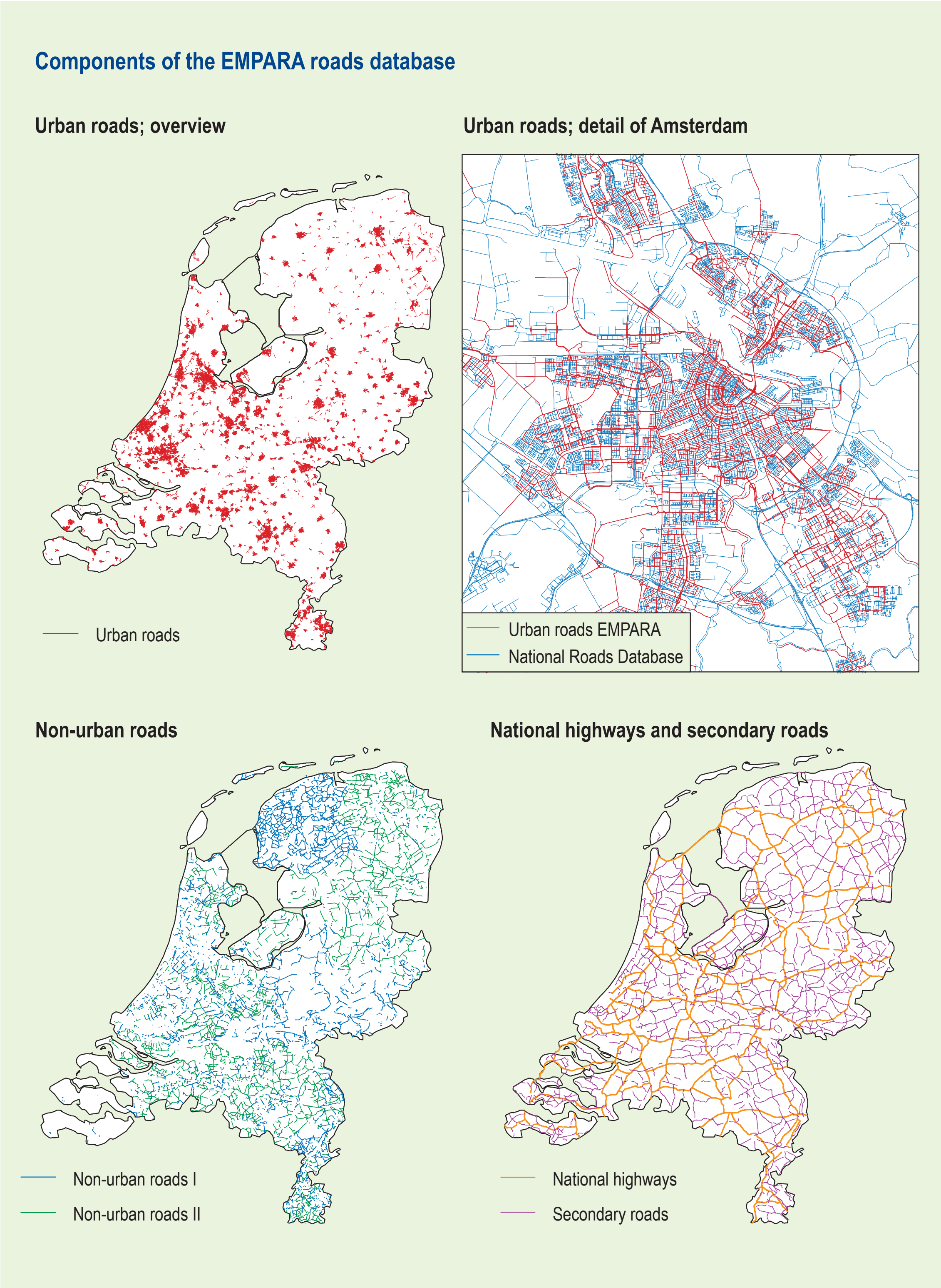

Figure 3.5 Components of the EMPARA roads database.

Clockwise from top left: a. Urban roads, overview of the Netherlands. b. Urban roads, detail of Amsterdam; the National Roads Database (NWB) are shown in blue, the roads used for EMPARA are shown in red. c. Non-urban roads; in blue, roads that correspond with the NWB, with great geometric precision (non-urban I, intensity > 5000 vehicles per day); in green, roads from the New Regional Models (NRM), shown schematically in a geometric sense (non-urban II, intensity < 5000 vehicles per day). d. Motorways and secondary roads.

required for noise calculations. For each train route, data about train traffic intensities, speeds and superstructure type are available, as well as the emission calculated from this data. This emission database is inputted directly into EMPARA.

For road traffic noise, the PBL uses EMPARA to calculate the emission based on traffic data, traffic composition, speeds, road surface type, etc. For this purpose, the emission formulas from SRM I are used. The emission factor for road traffic in the calculation regulations already contains a propagation component for a hard surface. In EMPARA, the emission factor is therefore corrected with a fixed reduction. This correction factor is 2 dB and is equivalent to a modification of the propagation term for ground attenuation.

The calculation and measurement regulations contain formulas for the emissions of light motor vehicles (passenger cars and light delivery vans) and heavy motor vehicles. These are based on large-scale measurements that were made in the 1990s. Heavy motor vehicles are divided into two classes: medium-heavy and heavy. For each vehicle category and for each 24-hour period, the emission is calculated based on the number of vehicles per hour (Q) and the speed (v). For each part of the day, the energetic sum of the emission factors is determined for each motor vehicle category; from this, a weighted emission is determined for the 24-hour period.

$$E_c = a_c + b_c \cdot \log \frac{v_c}{v_{0,c}} + 10 \cdot \log \frac{Q_c}{v_c} - 2$$

The parameters in this formula:

Category	A	B	v_0 [km/h]
Light vehicles	69.4	27.6	80
Medium-heavy vehicles	73.2	19.0	70
Heavy vehicles	76.0	17.9	70

The emission is determined for the standard road surface type: dense asphalt concrete. For other road types, a correction ($C_{roadsurface}$) is applied to the emission factor. For open graded asphaltic mixes, which are widely used in the Netherlands, the correction is -3 to -4 dB. For noisier road surfaces, such as brick paving, the correction factor has a positive value.

The calculation and measurement regulations are frequently used for road surface correction factors. These regulations contain correction factors based on measurements according to a fixed protocol.

As part of the monitoring programme of the PBL, a study was conducted of the most prevalent road surfaces on the major road network in the Netherlands (M+P 2007). This study, together with a study conducted by the Dutch road maintenance authority, showed that the actual correction factor for open graded asphaltic mixes is probably 3 dB (2 - 4 dB). See also Table 3.4.

Table 3.4 Correction factors for types of road surface.

Type of road surface	$C_{roadsurface}$
Fine (dense asphalt concrete)	0
Open graded asphaltic mixes	-3 dB, -4 dB
Brick paving	+4 dB
Concrete pavement	+2 dB

3.2.5 Noise propagation

This section describes the basic principles for noise propagation in the EMPARA model. First, the general propagation terms will be addressed, followed by the assumptions for more specific situations such as noise barriers and the barrier effect of buildings.

General, non-spectral modelling of noise propagation
EMPARA calculates the noise propagation (transmission) using a geometric dispersion term D_d, air attenuation D_{air}, a correction term for the absorption by the surface $D_{surface}$ and for weather conditions D_{meteo}. These four components are described in empirical formulas and can be added together. If sound barriers and/or buildings are present, an object attenuation term is added.

$$T = D_d + D_{air} + D_{surface} + D_{meteo} + D_{object}$$

The separate terms are made uniform for rail and road noise based on the definitions (which is different for road noise in the calculation and measurement regulations). However, the parameterization of the formulas for road noise and railroad noise differ regarding D_{air} en $D_{surface.}$ This is a result of the assumption that the sound spectrum for these sources is not the same.

The value of the transmission terms is shown in Figure 3.6.

The transmission terms for the general propagation of road traffic noise are shown below. The transmission terms for rail traffic can be found in the appendix.

The geometric dispersion of a line source is shown by the following:

$$D_d = -10 \cdot \log \frac{\theta}{\pi \cdot d}$$

where θ: the angle of sight with which the line source is viewed from the observation point, d: the perpendicular projection distance from the observation point to the line source.

$$D_{air,road} = 0{,}008 \cdot r^{0{,}85}$$

Attenuation and correction terms for road traffic noise

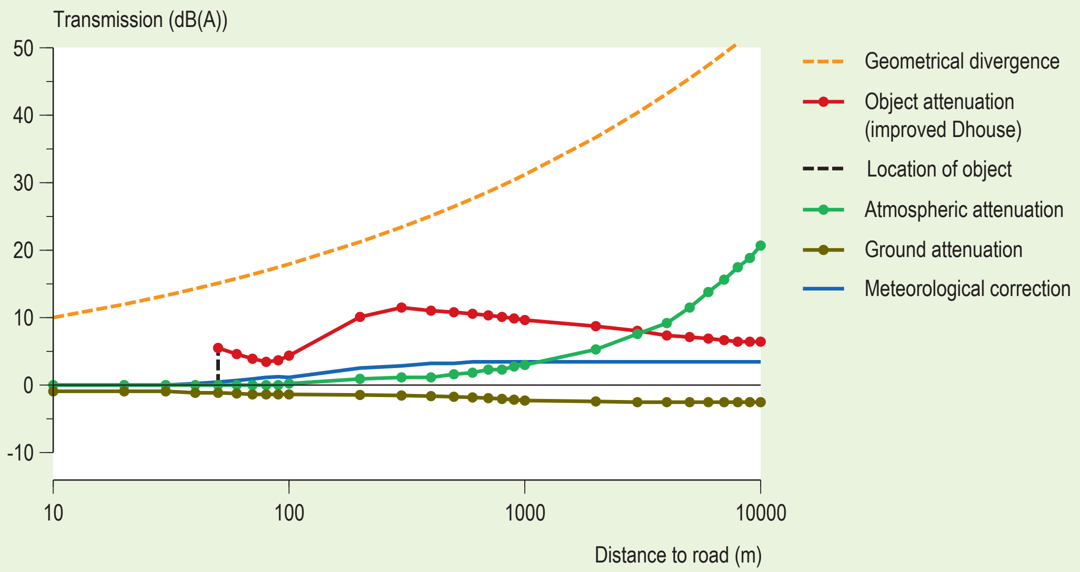

Figure 3.6 Distance-dependent attenuation and correction terms for road traffic noise.

Assuming a fixed observation height, the term $D_{surface}$ is calculated for roads as follows:

$$D_{surface,road} = 2(B-1) + 0{,}155B\left(1 - e^{-0{,}028r}\right)\left(1 + 15{,}84e^{-0{,}65h_{road}}\right) + 3(B-1)(1 - e^{\frac{-0{,}0065r}{h_{road}+5{,}4}}$$

Where B is: ground absorption

The first term in $D_{surface,\ road}$ is different than the definition in the calculation regulations. The emission factor has been adapted accordingly. The final term is an additional term for the area in the middle.

Weather conditions have a major influence on noise propagation, especially over large distances. The deflection of sound is determined by differences in sound propagation at various heights. This is determined especially by wind profiles. EMPARA, like SRM I, is based on a tailwind condition. This is a situation in which sound easily propagates and has a relatively large influence on the average annual noise level. The effect of weather conditions on noise propagation is indicated with the term D_{meteo} and is calculated as follows:

$$D_{meteo} = 3{,}5\ 1 - e^{-0{,}04\ \frac{r}{h_r + h_{road} + 0{,}5} - 2}$$

All other noise barriers and sound attenuating objects (forests, buildings etc.) are taken into account as additional attenuation, with the term D_{object}.

Table 3.5 Soil categories in the Land Use Database of the Netherlands with corresponding surface factors and building variables. The values in this table are currently based on rough estimates of the dimensions and densities of the objects and therefore provide only an indication of the expected effects. Statistical validation of the data in the table requires further research.

	Indication of soil category	B [-]	H [m]	$1/l_v [m^{-1}]$	[illegible] [-]
1	Grass	1	0	0	0
8	Greenhouses	0	3	0.02	0.5
9	Orchard	1	5	0.01	0.8
11	Deciduous forest	1	10	0.03	0.8
12	Coniferous forest	1	10	0.03	0.8
13	dry heather	1	0	0	0
14	open nature reserve with plant cover	1	5	0.002	0.8
15	bare soil in nature reserve	1	0	0	0
16	open inland water	0	0	0	0
17	open coastal water	0	0	0	0
18	urban built-up area	0			0.5
	very highly urbanized		20	0.02	
	highly urbanized		15	0.02	
	moderately urbanized		10	0.015	
	slightly urbanized		5	0.008	
19	buildings in countryside	0.5	5	0.004	0.5
20	deciduous forest in built-up area	1	10	0.03	0.8
21	coniferous forest in built-up area	1	10	0.03	0.8
22	forest with dense building	0.5	10	0.015	0.6
23	grass in built-up area	1	0	0	0
24	bare soil in built-up countryside	0.5	0	0	0
25	main roads and railroads	0	0	0	0
27	Agriculture	1	0	0	0
30	no data	0	0	0	0

No spatial information is available about the elevation of the roads and railways above ground level. In EMPARA, the calculations are conducted based on a fixed elevation for roads and railways. For motorways, a height of 1 m above ground level is assumed, for other roads, it is assumed that they are at ground level. Railways are often built on top of dikes; therefore a fixed height of 2 m is assumed.

Ground attenuation

This is defined as a function of the ground absorption B, i.e. the proportion of the surface between source and receptor that is soft. This factor is 0 for hard surfaces such as water, asphalt, concrete, brick paving etc., and is 1 for soft surfaces such as grassland, farmland, forest land etc. Data about surface condition in quadrants of 25 x 25 m² are known from satellite photographs (the Land Use Database of the Netherlands). In Table 3.5 a summary is provided of these soil conditions with a proposal for an accompanying value for B_k for a quadrant. Note that B_k can have the discrete values of 0, 0.5 or 1. The other units in the table will be discussed later on.

The surface fraction for the transmission path [4] between source and receptor is determined by a path-length weighted average of the ground fractions in all quadrants that are transected by the line connecting the source and receptor. The ground fraction B then follows from:

$$B = \frac{1}{L}\sum_{k=1}^{N} L_k \cdot B_k$$

where B_k is the fraction for ground absorption in quadrant k, L_k is the length of the transmission path within quadrant k, L is the length of the transmission path and N is the total number of quadrants that are transected by the transmission path. This averaging can also be used to determine values for areas larger than the 25 x 25 m² quadrants.

Modelling noise barriers

In the Netherlands, noise reduction is realized by means of noise barriers or earth walls, which are installed along more than 16% of the roadways. The effectiveness of a noise barrier is strongly dependent on the frequency spectrum of the noise source. Calculation methods that determine the attenuation are therefore best suited if they describe effectiveness for each octave or 1/3 frequency band. For the purpose of large-scale noise mapping. these methods are too detailed and time-consuming, so a simplified method for determining barrier attenuation is used in EMPARA. The formulas that are used to determine attenuation resulting from noise barriers near a source (line source or otherwise) are shown in Appendix 2.

Modelling noise propagation in built-up areas

There is no statutorily prescribed method to determine noise propagation behind the first line of buildings. In practice, the noise propagation for relatively small areas at some distance behind the first line of buildings is calculated with SRM II. This method places high demands on the spatial precision of the environmental characteristics. For larger areas, a method is used that is derived from the German VDI regulations. This method models both the protective effect of the first line of building and the sound transmission through the built-up area. During this process, the protective effect is calculated based on the so-called ‘acoustic detour’ which a beam of sound takes over the building. The transmission depends on the degree of openness of the built-up area, as this is “seen” by the direct sound beam. The total attenuation is then determined by the smallest of these two attenuation terms.

The idea behind the object attenuation model is that the noise level behind buildings is determined by two sound beams: a penetrating beam (which after being weakened by the first line of building undergoes additional object attenuation – effect DT) and a noise barrier beam (which, without additional object attenuation, goes from the top of the first line of buildings to the receptor point – effect Ds). The object attenuation results from the combination of both components in this 2-beam model using the method described by the formulas in Appendix 3. The contributions of a penetrating beam that undergoes scattering due to the building and of a barrier beam that results from the barrier of the first line of buildings are illustrated in Figure 3.7.

4) In the NOISTOOL software, *the average for B* over the transmission path between source and receptor is calculated with the ‘shifted transmission path’, i.e. the transmission path that links the middle of the source cell with the middle of the receptor cell.

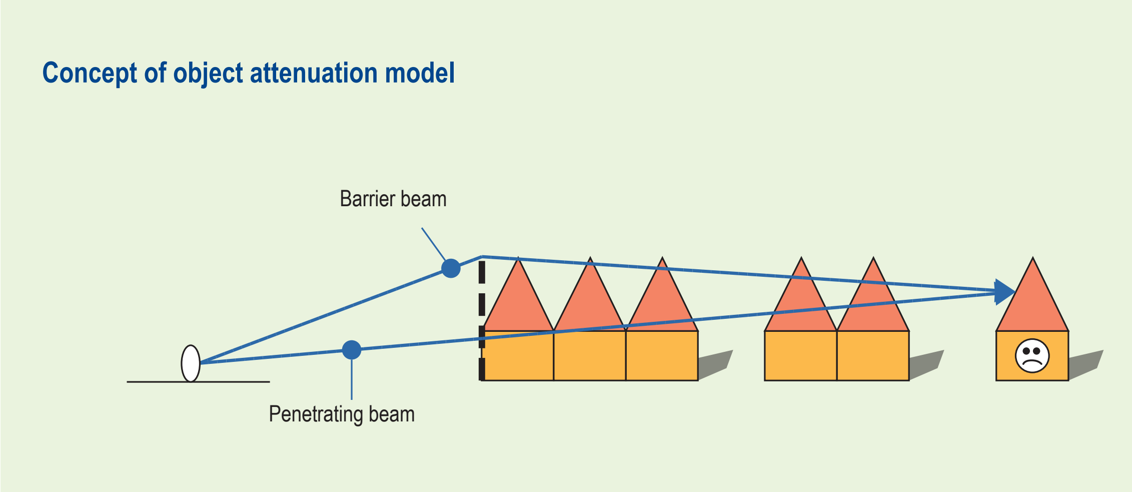

Figure 3.7 Model for Dobject: Barrier effect due to the first line of buildings and scattering of penetrating sound due to the buildings.

Determining the exposure of houses and residents

The team LOK has access to a database that includes all housing coordinates. This database was created from another database by removing all address coordinates in the Netherlands that are not houses. To this database, an average number of residents per house has been added. LOK does not have access to data concerning the actual number of residents per house, but this number is known in a 'postal code 6' area (comprising approximately 20 to 40 houses). This number is allocated to all houses in that area. In this way a reasonably good database has been acquired.

The noise value of the grid cell within which the house is located (the coloured areas in the above figure, grid cells of 25x25 m) is assigned to the database with the house coordinates (the points from the above figure). After this, a distinction is made by counting all the houses within a specific noise class or dB value.

In the same way, a distinction can be made by counting all the residents within a specific noise class or dB value.

3.2.6 Comparison of models

Deviations and uncertainties that are described with respect to the calculation and measurement regulations will also apply to EMPARA. The section on validation of models therefore also addresses the uncertainties in SRM I and II

As part of a study into the implications of the planned implementation of an EU directive involving noise mapping, an explicit comparison was made between EMPARA and the revised SRM II model for road traffic noise. Because this study was carried out in supplementation to the study in 2000 involving model comparison, the results have been included in the present report.

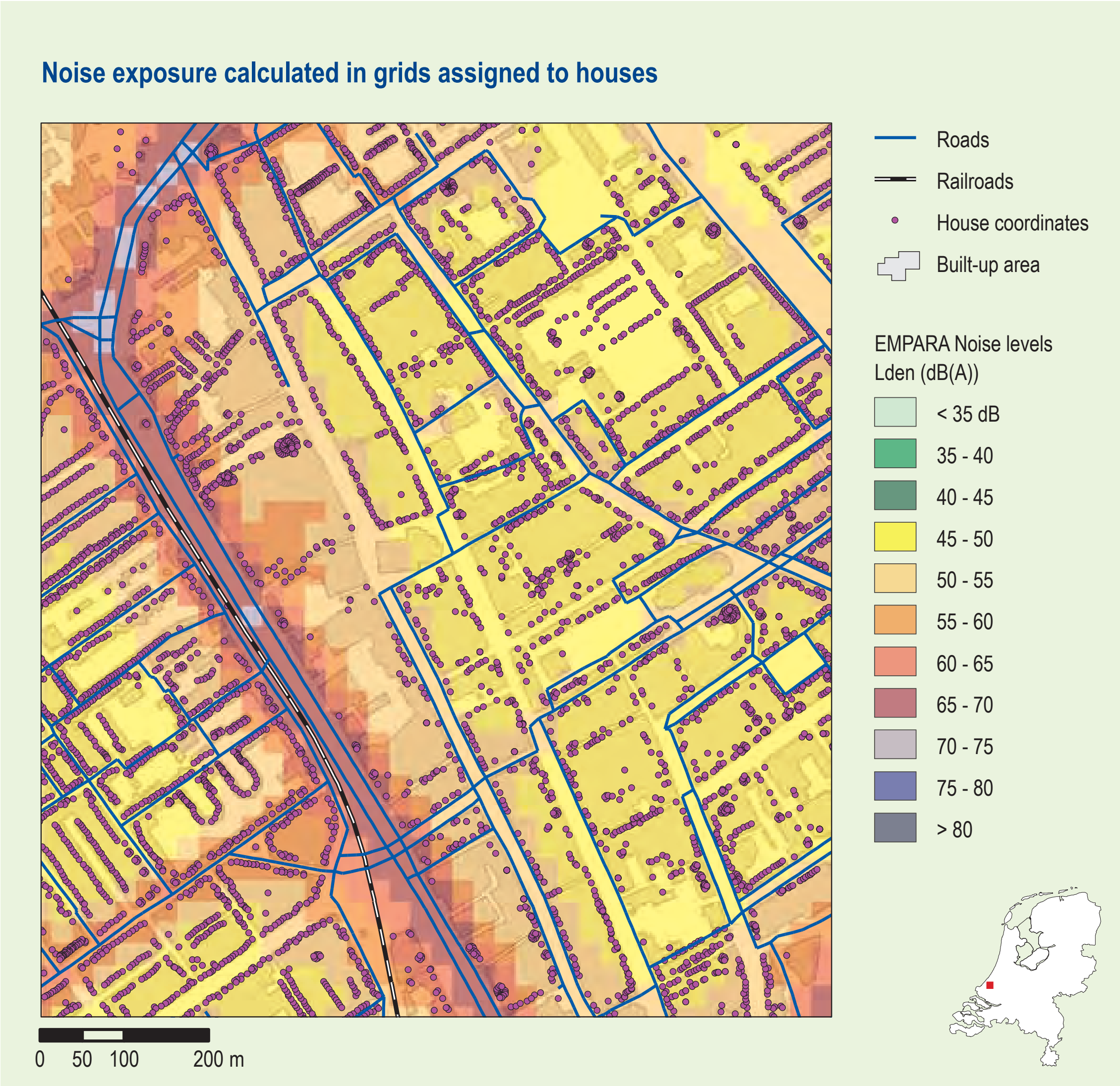

Figure 3.8 Detail of the map for noise level analysis.

The SRM II formulas used the concept of the revised Calculation and Measurement Regulations for road traffic noise 2006[5)] (www.stillerverkeer.nl). The EMPARA calculations are conducted with modified, improved modelling for barrier effects (see previous section) and object attenuation (due to buildings). A description of the modelling of emission and propagation is included in Appendix 1.

The comparison concerns only the influence of modelling the propagation, not the influence of the spatial discretization and schematization of the environment. The comparison is limited to the rural area (no buildings). This is because calculating the noise propagation over large distances in a built-up area is impossible with SRM II. The comparison was based on a situation where a single, infinitely long, straight road was modelled. At various distances from the road, the noise level was calculated. These calculations were conducted with and without a noise barrier at distances of 15 and 40 m from the road axis.

5) in Dutch: Reken- en Meetvoorschrift Geluidhinder 2006, Staatscourant 21 december 2006, nr. 249.

Figure 3.9 Comparison for national highway for hard and absorbent surface: with and without a 5 m noise barrier at an 18 m distance, road 1 m above ground level.

In the calculations, the following parameters were varied:
- road elevation (0 and 1 m)
- noise barrier height (0, 3 and 5 m)
- distance to noise barrier (10 and 20 m from road axis)
- ground attenuation (surface factors 0 and 1)

Two observation heights were modelled: 5 m (prescribed in the Noise Abatement Act) and 4 m (the observation height from the draft EU directive). The comparison also looked at the influence of traffic composition and speed. Traffic composition and speed effect noise emission.

The results of the calculations are shown in Figure 3.9 through Figure 3.11. In the figures, the results from EMPARA are shown with the abbreviation "LBV". The results show that the differences remain limited to between 1 and 2 dB(A), both with and without noise barriers. The LBV method produces a higher result of 3 to 4 dB(A) only for the situation involving a motorway with a 100% hard surface across the entire transmission path and a noise barrier.

Comparison LBV - SRM2 for secundary roads

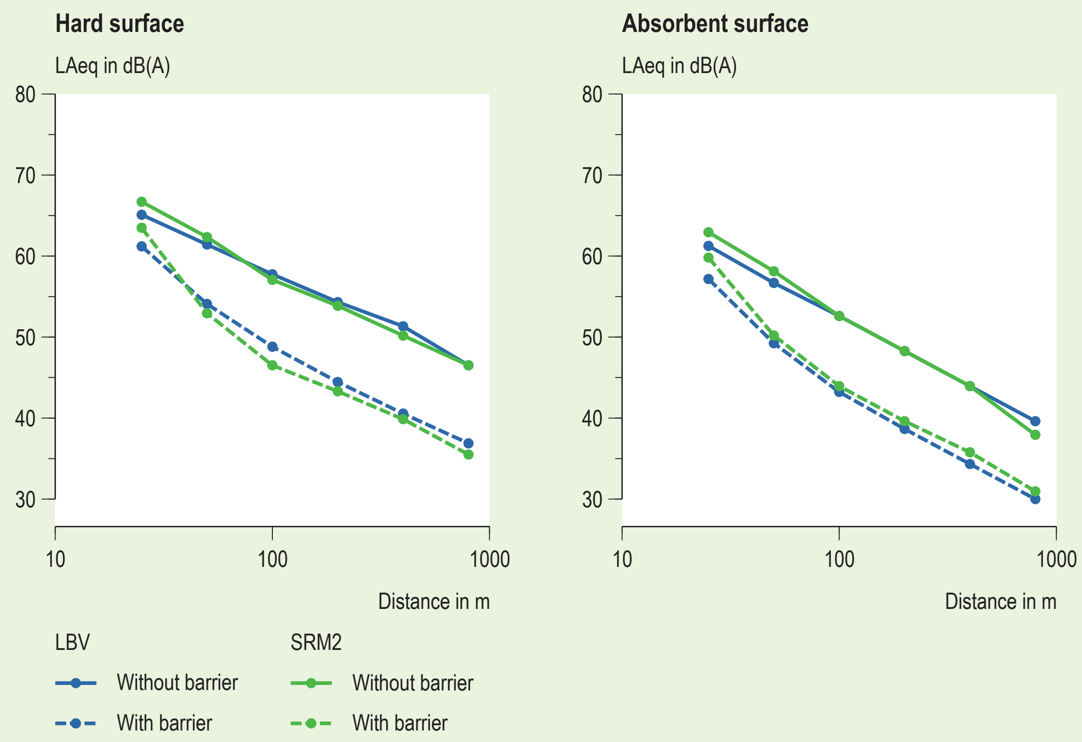

Figure 3.10 Comparison for a secondary road for hard and absorbent surface: with and without a 3 m noise barrier at a 12.5 m distance, road at ground level.

3.2.7 Validation of noise propagation modelling

The need to model beyond standard distances

For motorways, standard calculations are conducted to a distance of 2.5 km from the road; for secondary roads, a distance of 1.5 km is used. The models on which EMPARA is based were never developed and validated for large distances.

Calculations have shown that with standard calculation distances, the limit for the indicator for noise abatement areas is exceeded. Especially for motorways the noise level at the maximum calculation distance often turns out to be higher than the required 40 dB(A) $LAeq_{24uur}$ (≈ 36 dB(A) L_{den}). Figure 3.12 shows that this is the case in 60% of the total roadlength of motorways. Moreover, the lower limit for calculating annoyance (approximately 45 L_{den}) cannot be determined adequately in allmost 15% of the roadlength of motorways.

Previous research (Dassen, 2001) has shown that the urban area is insensitive for increasing the calculation distance. Only for motorways within urban areas, increasing the maximum calculation distance does have some effect on the calculated area affected by levels between 50 and 65 dB(a) and thus on the percentage of residents that is exposed to this noise level. However, the sensitivities are limited to a maximum of 3%.

Figure 3.11 Comparison for an urban road for hard and absorbent surface: with and without a 2 m noise barrier at an 8 m distance, road at ground level.

The limitations due to maximal distance led to the conclusion that extending the range of the model was necessary. As this extension leads to calculations far beyond the formal application area of the standard methods, TNO TPD was requested to make a comparison with a numeric model with which reliable calculations can be conducted over large distances. This parabolic equation model (PE) was developed to calculate the effects of a non-homogeneous atmosphere on sound propagation and meanwhile has been validated and used for a number of studies in this area (Dassen, 2001).

Validation of modelling up to 2 km

In 2000, a validation is initially conducted at distances up to 2 km. The comparison is based on a sound emission spectrum that is reasonably equivalent to the standard road traffic spectrum. The surface is assumed to be grass. With calculations using the PE model, the acoustic impedance of the surface can be varied within this type. Two variations are used in this comparison. With the PE model, an arbitrary tailwind profile can also be calculated. In this case, the profile that is most compatible with the tailwind situation of the standard calculation method has been chosen. For the comparison, SRM II is also used. This implies the choice of a ground factor (B) of 1 to indicate soft ground (grass). For a more detailed description of the comparison, see Gerretsen (2005). The results of the comparison are shown in Figure 3.12.

The figure shows that up to the maximum calculation distance of 2 km, there are no large differences between the various models. At larger distances, the EMPARA model results in somewhat

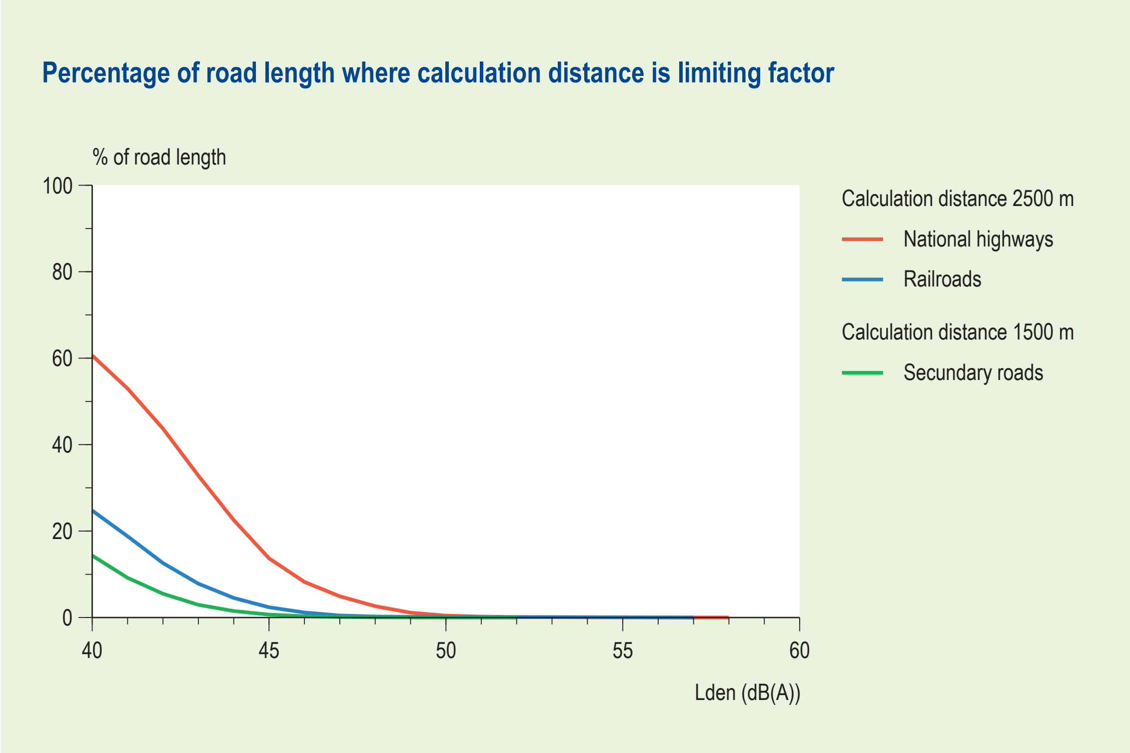

Figure 3.12 Exceeding of noise levels at the maximum calculation distances for motorways, secondary roads and railroads.

higher levels than those that are calculated with SRM II (up to 3 dB(A)). A comparison with the PE results shows that the values for propagation given by the EMPARA model lie within the PE values for almost all distances. Depending on the chosen surface impedance, the difference with the PE values is no more than 2 dB(A).

Note that TNO qualified the results as follows:

i) Previous research has shown that a completely hard surface can yield levels that are much higher than those provided by the standard calculation methods, and therefore by EMPARA as well. This can have an effect in situations where much of the surface is hard (for example near large bodies of water).
ii) This comparison did not consider the influence of noise barriers. At the time the comparison was made, there were indications that the effect of wind on noise barriers is greater than that assumed in the standard calculation method, and also greater than that in the improved modelling in EMPARA.

Validation of modelling up to 8 km

In 2007 the validation is extended to distances greater than 2 km. This is possible because the PE model is validated against the measured noise from heavy weapons at distances up to 10 km.

The second validation is generally structured in the same way. The results from EMPARA are compared with those from SRM I and the PE model. However, in the second validation a different approach is taken to meteorological influences. In the PE model, the prevailing meteorological situations in the Netherlands are converted into 27 sound velocity profiles with statistical weights indicating how often a profile occurs.

Noise level as calculated by different models

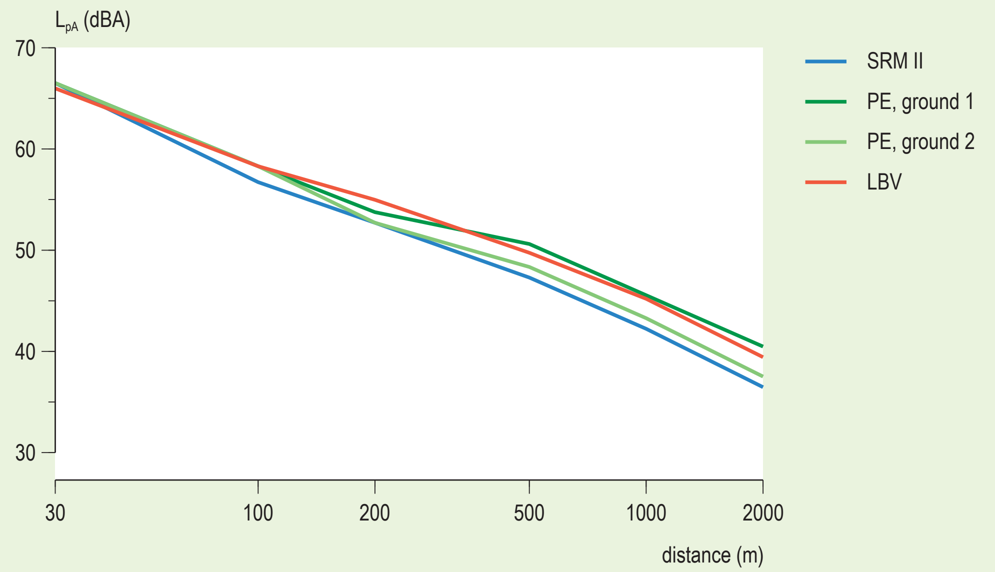

Figure 3.13 Noise level in dB(A) at 5 m elevation above a flat grass surface, assuming a tailwind situation, as a function of the distance to an infinitely long, straight road, according to SRM II, the PE model (2 different ground impedances) and EMPARA (shown with the abbreviation LBV) (source TNO).

Figure 3.14 shows that the differences between EMPARA and PE are very limited. The results from EMPARA at 2 km and 5 km distances are at the top of the bandwidth of the results of the PE model. At the 8 km distance, the EMPARA result is 1 dB below the lowest of the corresponding results from PE. If the results of the PE model are averaged across the four different orientations of the observation point, the results of EMPARA at 8 km distance is 2 to 3 dB below this average (depending on the ground type that is assumed in the PE model). The figure also shows that as the distance to the road increases, the results from SRM I increasingly diverge from the results of EMPARA and PE. At a distance of 8 km, SRM I provides a 10 dB lower noise level Lden. This comparison shows a good correspondence between the noise propagation calculated with both PE and EMPARA for large distances above open areas.

A qualification to this comparison is that the validation was conducted for noise propagation in a non-built up area with an absorptive surface. On this basis, a correspondence between the model results cannot be expected for noise propagation in urban areas. The influence of buildings is especially large when they are located near the noise source (the road) and/or the observation point. If the road and the observation point are both located in an open area, then the influence of scattered buildings (villages) between the road and the observation point is smaller.

Validation of the effect of noise barriers

Model results of EMPARA were compared with measurements (Swart, 2002). The measurements were conducted at three different sites on two barriers and on an earth wall. At both barrier locations, the model attenuation is in good agreement with the measured values, apart from one position in the far shadow region of the barriers, where the model overestimates the attenuation

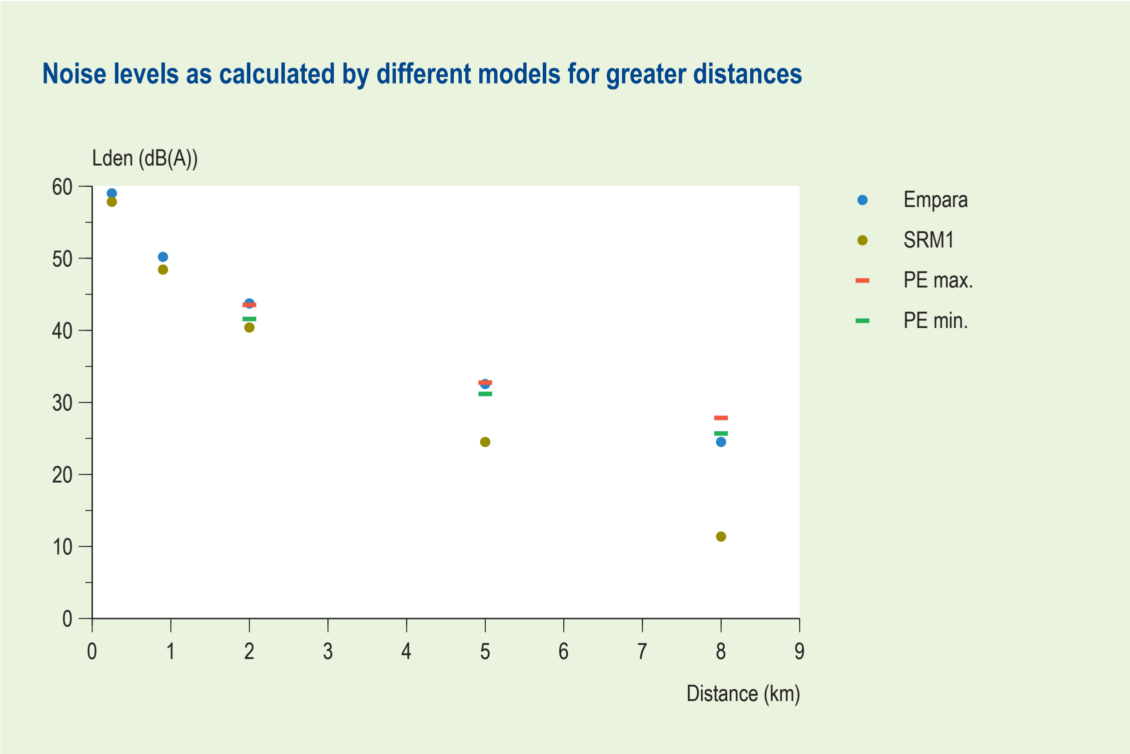

Figure 3.14 Noise level Lden as a function of the distance to a motorway, calculated with SRM I (in accordance with the calculation and measurement regulations for road traffic noise), EMPARA and PE (PE only for 2, 5 and 8 km distances).

by 4 dB(A). Compared to a more elaborate spectral method, as required by Dutch noise legislation (SRM II), the differences remain within 2 dB(A).

3.2.8 Sensitivity analysis

Sensivity of emission
The emission depends on three factors:
1. Traffic intensity
2. Distribution of vehicle types
3. Traffic speed

1. Traffic intensity
The sensitivity for intensities in the model is limited by the way in which the noise values are calculated. Due to the logarithmic scale, a limited increase or decrease of vehicle numbers has only a limited effect on the emission, and therefore also on the immission.

2. Distribution of types of vehicles (light, medium-heavy and heavy vehicles)
The calculation and measurement regulations, and therefore EMPARA as well, take account of differing noise levels from different types of vehicles. The vehicles are consequently placed into three classes. The distribution between the classes determines the result of the calculation.

For motorways, this distribution is well known; for secondary roads there is reasonable data based on a 2001 survey. However, for municipal roads, the values are not available.

Table 3.6 Sensivity of the noise model for deviations in traffic intensity.

Traffic reduction		Traffic increasement	
10 %	-0.5 dB(A)	20 %	+0.8 dB(A)
20 %	-1.0 dB(A)	40 %	+1.5 dB(A)
30 %	-1.6 dB(A)	60 %	+2.0 dB(A)
40 %	-2.2 dB(A)	80 %	+2.6 dB(A)
50 %	-3.0 dB(A)	100 %	+3.0 dB(A)
75 %	-6.0 dB(A)	300 %	+6.0 dB(A)

Modifications of these values can therefore lead to a higher or lower emission from the road, and therefore to a different noise load on the houses and residents.

Table 3.7 shows the normalized measurement values for the various types of vehicles from the calculation and measurement regulations. This shows that shifts in the proportions of vehicles can lead to significant increases of up to 6.6 dB (if there is a shift from 100% light vehicles to 100% heavy vehicles).

3. Speed
The speeds that are used in the calculations for the various road types (urban traffic, non-urban thoroughfares, motorways) are stipulated in the regulations. In some cases, different speeds will apply to actual situations, or speeds will be driven that deviate from the legal speed limit on the road. On average, however, the deviations during one year are limited. The extreme values are then cancelled out.

Table 3.8 is an example of how much the emission will deviate if a lower speed is driven than is assumed in the model. At higher speeds, the reversed values apply.

Sensivity of propagation for road elevation and ground absorption
Due to the lack of information about the elevation of roads, in the EMPARA calculations it is assumed that motorways have a standard elevation of 1 m above ground level. For secondary roads and urban roads, it is assumed that their elevation is equal to ground level. For railroads, a standard elevation of 2 m is assumed.

The hardness of the surface is estimated based on the national land-use database, *Het derde Landgebruik Nederland* (LGN3). In this way, the LGN categories 'greenhouses' and 'open water' are assigned a ground absorption factor of 0 (acoustically hard), while 'grass' and 'forest' are assigned ground absorption factor of 1 (acoustically soft).

Table 3.7 Sensivity of the noise model for deviations in vehicle type.

Category	A (dB)	Increasement (dB)
Light vehicles	69.4	
Medium-heavy vehicles	73.2	+ 3.8
Heavy vehicles	76.0	+ 6.6

Table 3.8 Sensivity of the noise model for deviations in traffic speed.

Speed reduction (10% heavy traffic)	
From 110 to 100 km/h	-0.7 dB(A)
From 100 to 90 km/h	-0.7 dB(A)
From 90 to 80 km/h	-1.3 dB(A)
From 80 to 70 km/h	-1.7 dB(A)
From 70 to 60 km/h	-1.8 dB(A)
From 60 to 50 km/h	-2.1 dB(A)
From 50 to 40 km/h	-1.4 dB(A)
From 40 to 30 km/h	0 dB(A)

Dassen (2001) shows that the effect on the noise level of using standardized road elevations and surface absorption factors is limited to a few dBs. A difference of more than 3 dB at large distances (500-1000 m) is produced only with a 2 m road elevation in the urban area. This is caused primarily by the reduced noise barrier effect of the first line of buildings.

Sensitivity of indicators for emission and propagation.
PBL previously conducted research into the sensitivity of several indicators. In this section, the results will be summarized. The sensitivity of the surface area and the number of residents in a specific exposure class for a generic variation in the noise level were investigated. In this study, the noise level was changed uniformly at all locations. This is representative of structural deviations, so these are worst cases. The sensitivity is determined by changing the emission by various gradations.

Table 3.9 shows the relative sensitivity of the indicator values for generic changes in the noise level, e.g. as a result of change in emission. The relative sensitivity, SΔ, of the indicators for a generic change in the noise level, D, per decibel change, is defined as:

$$S \;\; = \frac{I \;\; - I_{nom}}{I_{nom}} \cdot \frac{1}{\;}$$

where I_{nom} is the indicator value that follows from the noise level as it is calculated with the nominal input data and settings. I is the value that is obtained after a generic change of this noise level.

Table 3.9 Relative sensitivity of indicators for generic changes in the noise level, per decibel of change. The left column contains the values with which the noise level changes.

	Residents 51-65 dB(A)	Residents > 65 dB(A)	A > 50 dB(A)	% annoyance	% serious annoyance	Norm exceedance in noise abatement areas
-3	-0.03	-0.2	-0.04	-0.05	-0.08	
-1	-0.07	-0.4	-0.09	-0.1	-0.08	
+1	0.04	0.6	0.09/0.08	0.1	0.2	0.3
+2	0.03	1.4	0.02	-	-	
+3	0.02	0.3	0.04/0.1	0.05	0.1	0.3

Example: If the sensitivity of the noise-exposed area is assumed to be 0.3, this means that if the estimated emission is too low by 1 dB(A), then an increased noise emission of 1 dB(A) will cause this indicator to increase by 30%.

The table shows that the number of residents exposed to a noise level higher than 65 dB(A) is significantly more sensitive for inaccuracies in the noise level than the other indicators. For a calculation that includes the entire Netherlands, the sensitivity can increase to 0.6 per dB. In fact, the calculations for the town of Leiden show a sensitivity of 1.4 per dB

This sensitivity implies the following: if there is a measurement uncertainty in the calculation of the noise level of 1 dB, an uncertainty which is higher than this value must be taken into account when calculating the number of residents exposed to a noise level above 65 dB(A). The explanation for this large sensitivity lies primarily in the fact that the relatively high levels of 65 dB(A) always occur at locations near the roads (up to several tens of meters). The first line of buildings often begins at these locations. A relatively small change in the noise level can lead to a relatively large shift in the number of houses exposed to noise.

The sensitivities of the other indicators are between several percent to several tens of percent per dB. The number of residents exposed to a noise level between 50 and 65 dB(A) is the least sensitive for inaccuracies in the calculation of the noise level (a maximum of 7% per dB).

For that matter, the table shows that the sensitivities have a non-linear correlation with the generic noise level variation. This is shown by the fact that the sensitivities depend on the amount and the sign of the generic change. This means that the precision not only affects the uncertainty in the absolute indicator values, but also the relative trend values.

The sensitivities from Table 3.9 can be used to estimate the magnitude of the effect on the trends for road traffic noise. According to the *Environmental Outlook* 5, the noise level from road traffic is expected to increase by more than 2 dB(A) in 2030. This increase will almost double the number of residents exposed to high noise levels (> 65 dB(A)). This is compatible with the values in the data table of 0.3 (at a 3 dB generic increase) and 0.6 dB (at a 1 dB generic increase) per dB for the sensitivity of the number of residents exposed to a noise level above 65 dB(A). However, if the noise level across the entire Netherlands was overestimated by 1 dB (half of the assumed maximum measurement uncertainty in the average noise level) for the reference year, then the exposure (> 65 dB(A)) in the reference year would only amount to approximately 60% of the currently calculated value. [check] If the noise level increased by 2 dB, this would mean an increase of nearly 170% (up to 160% of the value now calculated for the reference year). In this example, the sensitivity of the trend values for measurement uncertainty in the calculated noise level is therefore virtually the same as the sensitivity of the absolute values (also 70% with a systemic error of 2 dB). In many cases, however, the sensitivity of the trends is smaller than that of the absolute values.

3.2.9 Uncertainties

Uncertainty in the data

The data for motorways are maintained with very high accuracy by the Directorate for Public Works and Water Management. Deviations in the data will therefore be minimal. The intensities are measured at multiple locations on the motorways; the intervening sections are filled in based on measurements taken on the surrounding sections and during previous years. The proportions of light, medium-heavy and heavy vehicles are known according to the same method. For motorways, the 24-hour distribution is correctly processed in the input data.

Measured speeds are also known, but in accordance with the calculation model have been set to standardized values. There are only limited deviations within the annual average.

Data about road surfaces and noise barriers are also precisely maintained in a similar fashion.

The information about secondary roads originates from an inventory conducted in 2000/2001. For the years after this, the vehicle numbers have been increased by an average annual growth in traffic for all roads. Deviations from the national average due to changes in a local situation therefore result in a deviation in emissions from the road. These deviations are on the order of those shown in Section 3.2.8. The remaining data were obtained from an inventory in the same year and have not been updated since. Deviations from these values will lead to different noise emissions. For the 24-hour distribution, a national estimate is used.

Measured speeds are also known, but in accordance with the calculation model have been set to standardized values. There are only limited deviations within the annual average.

Installing additional noise barriers or noise-reducing asphalt will therefore lead to a deviation from the model calculations.

The data from municipal roads have been empirically determined based on information from 200 municipalities from 1990-1995, where equations were established for various types of roads and numbers of residents of the municipalities. These equations were applied to all other municipal roads. For the years after this, the vehicle numbers have been increased by the average annual growth in traffic for all roads. Deviations from the national average due to changes in a local situation therefore result in a deviation in emissions from the road. These deviations are on the order of those shown in Section 3.2.8. For the 24-hour distribution, a national estimate is used.

Measured speeds are also known, but in accordance with the calculation model have been set to standardized values. Deviations can be significant because on this type of road there are many intersections and other speed-limiting factors. On the other hand, a speed of 50 km/h is assumed, which often means an increase in the emission.

The proportions of light, medium-heavy and heavy vehicles are not known, and have been estimated for the various road types based on relevant literature.

For the municipal roads, there was no access to information about noise barriers or road surface types. Possible traffic-calming measures on and around a road are also not included. At a very localized level, this can lead to significant deviations. An example of this is shown in Table 3.10 with approximate values.

An inventory of rail traffic is made annually by Prorail, which is available. Virtually no deviations will be present in the data.

Table 3.10 Effects of traffic management measures on noise reduction.

Traffic management measure	Potential noise reduction (LAeq)
Traffic calming / Environmentally adapted thoroughfares	Up to 4 dB(A)
30 km/h zone	Up to 2 dB(A)
Roundabouts	Up to 4 dB(A)
Round-top/circle-top road humps	Up to 2 dB(A)
Speed limits combined with signs about noise disturbance	1 - 4 dB(A)
Night time restrictions on heavy vehicles	Up to 7 dB(A) at night
Rumble strips of thermoplastic	Up to 4 dB(A) noise increase
Rumble areas of paving stones	Up to 3 dB(A) noise increase
Flat-top humps	Up to 6 dB(A) noise increase
Narrow speed cushions	Up to 1 dB(A) noise increase
Rumble wave devices	0 dB(A)

Modelling in complex sound propagation situations

The acoustic advisory bureau DGMR has published reports on the differences between the methods as they are observed in specific situations. During this process, the results of different versions of “Method 1.5” were compared with each other. Based in this comparison, the uncertainty of Method 1.5 was estimated. The magnitude of this uncertainty was determined based on the relative standard deviation shown by the various versions of Method 1.5 as they are in use, with respect to each other and/or based on the deviation with respect to Method II. Because both the relative standard deviation and other deviations are directly related to the chosen approaches, this can lead (also with averaging over a large number of comparable situations) to systematic deviations from the actual values. Table 3.11 provides a summary of the uncertainty which must be taken into account when applying “Method 1.5”.

Uncertainty due to grid size

esides the precision of the noise level calculation, the discretization and the related schematization also affect the model results. The spatial discretization is established by calculating on a grid. As a result, values can be assigned to the model parameters only at the source and reception points. This choice of the values is therefore adapted to the average situation in the cell of the corresponding grid point. The calculated noise level must also be interpreted as an average value for the entire cell, and not as the noise level at the location in the middle of the cell.

Table 3.11 Main uncertainties regarding the use of th e “Method 1.5”-approach.

Situation	Uncertainty (in dB(A))	Importance
Noise barriers along roads or railroads	7	Relative standard deviation[1]
Propagation with varying surface type	3-5	2 dB relative standard deviation[1], ± 5 dB(A) maximum deviation[2]
Situations with a noise barrier where the reflection contribution is indicative	3-20	17 dB relative standard deviation[1], ± 20 dB(A) maximum deviation[2]
Open graded asphaltic mixes combined with noise barriers	2	Deviation[2] (systematic overestimation)

[1] Mutual deviation in the results of different versions of Method 1.5.
[2] Deviation with respect to the result from SRM II

Table 3.12 Deviation from a natural profile within grid cells of 25 and 100 m, at various distances from a source.

Distance(m)	Grid size	Higher or lower values than the calculated centroid			
200-300	100*100	+1.47 / -0.95			
	25*25	+0.36 / 0.32	+0.32 / 0.29	+0.29 / 0.26	+0.26 / 0.24
900-1000	100*100	+0.38 / -0.36			
	25*25	+0.09 / -0.09	+0.09 / -0.09	+0.09 / -0.09	+0.09 / -0.09
1900 - 2000	100*100	+0.23 / -0.23			
	25*25	+0.06 / -0.06	+0.06 / -0.06	+0.06 / -0.06	+0.06 / -0.06

However, by varying the cell size, input data can be provided with greater precision. For example, the land-use map LGN3 is available on a 25 m grid. Calculations with larger grid cells leads to aggregation of the information, and therefore to loss of precision. Smaller grid cells will not add additional information, and therefore only lead to a limited improvement.

The average value of a grid cell can deviate from the value that can possibly be calculated at the edge of the cell. Because houses can be at random locations in a cell, the value that is assigned to a house as part of exposure calculations (the centroid of the grid cell in which the house is located) can deviate from the value that would be calculated if the actual noise level on the house was determined. In that respect, the smaller the grid size, the more realistic th espatial distribution of the noise levels.

The deviation is shown in Table 3.12 based on an example calculation. The deviation is also shown in Figure 3.15 and Figure 3.16 for purposes of clarity.

The figures below show a number of 100 m grid cells, subdivided into cells of 25 m, with the corresponding calculated values. It is obvious that the possible deviations at the edge of the grid cell are significantly smaller if the cells themselves are smaller.

In addition, it is notable that the differences between the cells near the source are many times larger than the differences further away from the source. Reducing the size of the grid cells will therefore have a large effect, especially on the exposure distribution of the houses with the highest noise level (>65 dB).

For the inner urban area, there is a maximum sensitivity of -16% (number of residents exposed to levels above 65 dB(A)) for the grid size (Dassen, 2001). However, this sensitivity decreases if the noise level increases. This is because the increased noise level leads to an increase in the number of houses exposed to a noise level above 65 dB(A). As a result, the sensitivity for the grid size becomes relatively lower. As can be expected, the grid size only has a marginal effect on the results for the national area when determining the sound level in the National Ecological Network and noise abatement areas (>40 dB(A) $LAeq_{24h}$).

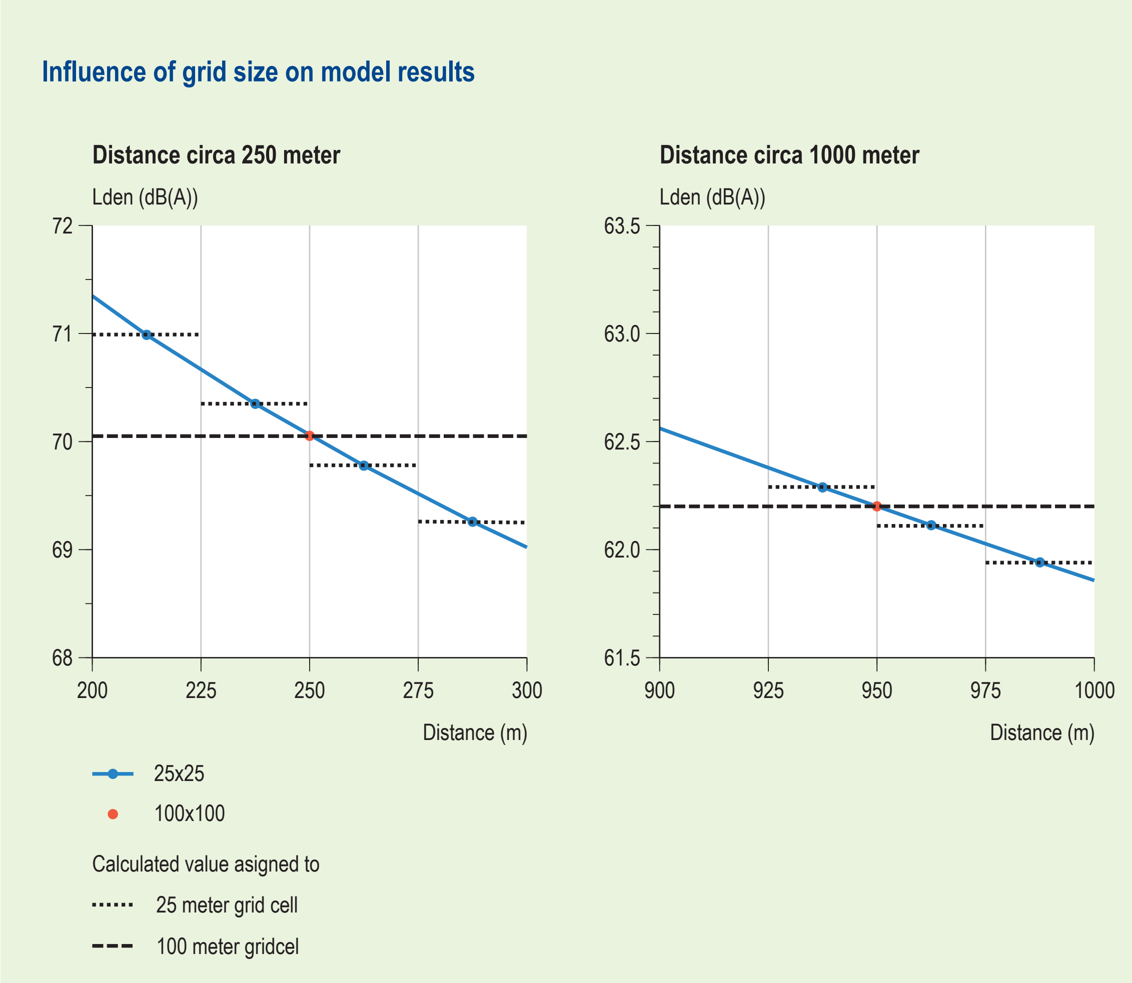

Figure 3.15 Deviation from natural profiles within grid cells of 25 and 100 m, at distances from the source of 200 m (left) and 100 m (right).

3.3 Air quality

3.3.1 Indicators

European legislation has established quality targets for outside air. These targets have been included in Dutch legislation. The air quality policy in the Netherlands aims to comply with these targets and to make the health risks from air pollution as small as possible. The indicators for evaluating air-quality policy focus on these aims. The indicators are established for NO_2 and PM_{10}, for both the past year and for future prognoses. Prognosis years that are important to current policy are 2010 (2011) and 2015. These are related to the years during which the air quality standards for PM_{10} and NO_2 will go into force, assuming that the Netherlands is granted a postponement to comply with the requirements. For longer-term prognoses, incidental calculations have also been made for 2020 and 2040

The basis indicator is a map of the Netherlands, made up of 25 x 25 m grid cells, with the concentrations of air pollutants. As an example, Figure 3.17 shows the map with the yearly average concentration of NO_2 in 2006.

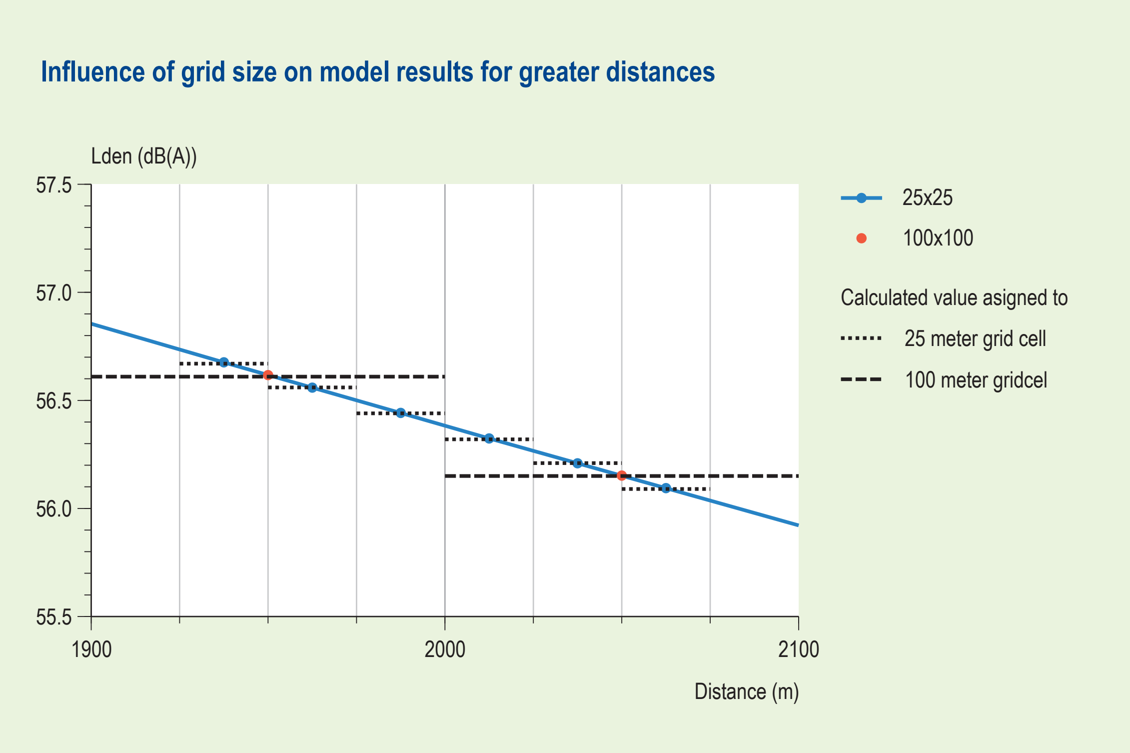

Figure 3.16 Deviation from natural profiles within grid cells of 25 and 100 m, at distances from the source of app. 2000 m.

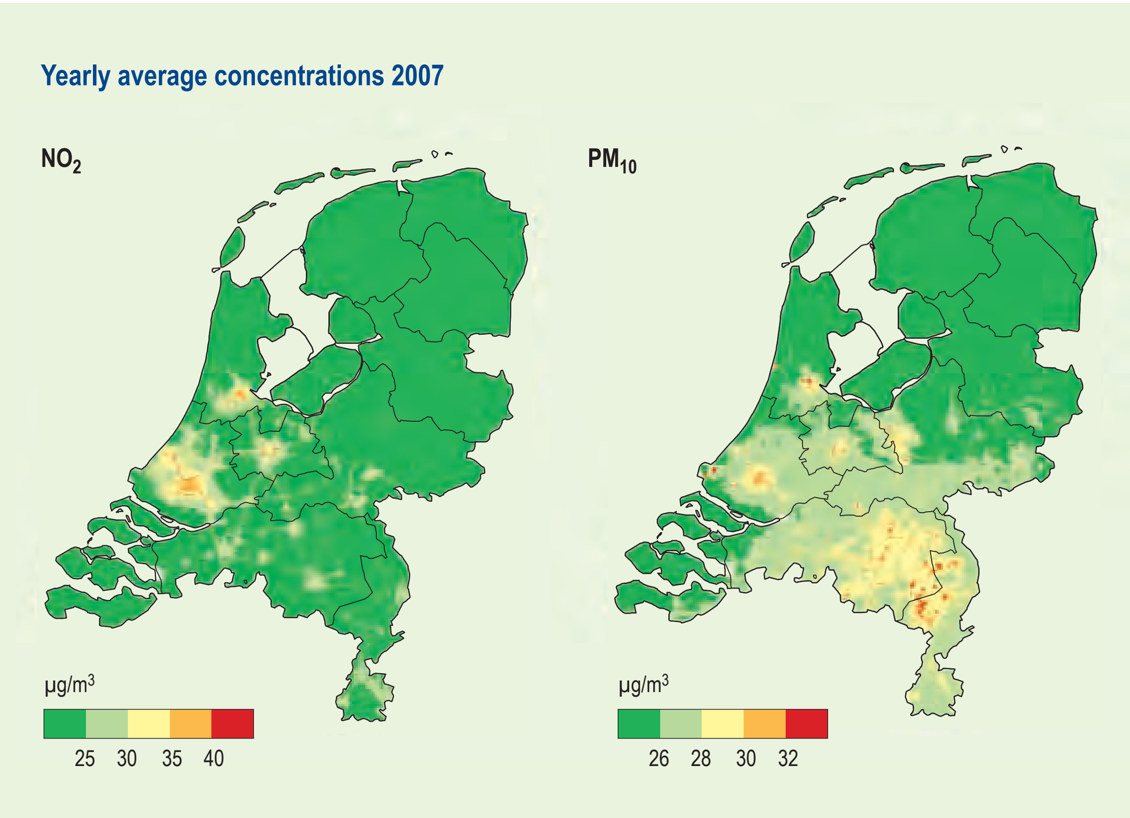

Figure 3.17 Yearly average concentrations of NO_2 and PM_{10} in 2007. The maps are the result of calculations with Luvotool.

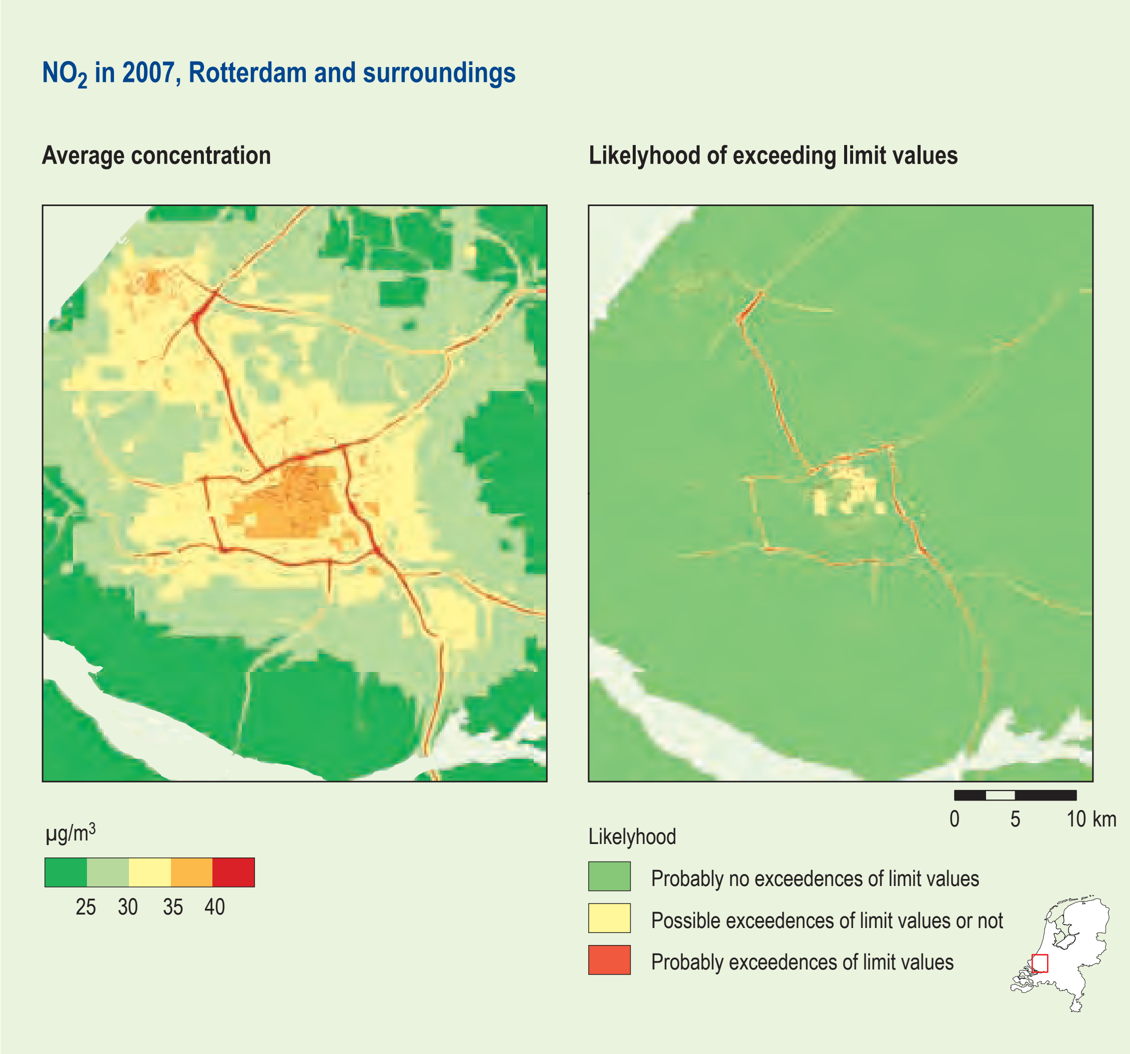

Figure 3.18 Detail of the concentrations map (Rotterdam and surroundings) for NO_2 (left) and map with likelihood of exceeding the limit values (right).

The other policy indicators for air quality are derived from the map:
– norm exceedences in cities (in km of road length)
– norm exceedences along motorways (in km of road length)
– exposure of the Dutch population to air pollution, divided into classes of concentration levels.

The maps in Figure 3.17 show the results of the calculations without taking into account the incertainties in the results. To do justice to the uncertainties a way of presentation is developed that reflects the likelihood of exceeding the limit values. Figure 3.18 shows an example of this in comparison with the concentrations map.(See also Chapter 4 for a discussion on this.)

Sea salt correction for suspended particles (PM10)

The European directive and the Netherlands air quality act that is based on the EU legislation allow a correction of the concentration of PM_{10} for particulate matter of natural origin if this does not contribute to the health hazard of the particulate matter. For the Netherlands, sea salt air aerosol is important in that case. The proportion of sea salt in the yearly average concentration of suspended particles (PM_{10}) ranges between approximately 7 µg/m3 along the west coast to approximately 3 μg/m³ in the eastern region of the Netherlands. To determine the yearly average concentration that is corrected for sea salt, a location-dependent correction is required. The Air Quality Evaluation Regulation (AQER) defines this correction for each municipality.

For determining exceedances of the 24-hour average concentration of suspended particles (PM_{10}), the above correction does not apply. It turns out that the effect of the concentration of sea salt in the outside air on the number of days on which the concentration of suspended particles exceeds the value of 50 μg/m³ is virtually the same throughout the Netherlands. The number of days on which the 24-hour average exceeds the limit value of 50 micrograms per m³ can therefore uniformly be corrected by reducing this number with 6 days.

Determining exceedances of the 24-hour average limit values for suspended particles (PM_{10})

The air module of EMPARA, as with most air quality models, calculates the yearly average concentrations of air pollutants. Other indicators must be derived from the yearly average. For PM_{10}, a relationship was derived for the Netherlands between the yearly average concentration and the number of days that the 24-hour average limit value of 50 μg/m³ for PM_{10} is exceeded. This relationship is described in the AQER (Staatscourant, 2007).

If the yearly average concentration $PM_{10} > 31.2$ μg/m³, then:

$$OD_{PM_{10}} = 4.6128 \times C_{yavg} - 108.92$$

If the yearly average concentration is 16 μg/m³ < PM_{10} < 31.2 μg/m³, then:

$$OD_{PM_{10}} = 0.13401(C_{yavg} - 31.2)^2 + 3.9427(C_{yavg} - 31.2) + 35$$

C_{yavg} = yearly average concentration PM_{10}

$OD_{PM_{10}}$ = number of days that the 24-hour average concentration of PM_{10} is higher than 50 μg/m³

According to the above relationship, with a yearly average PM_{10} concentration of 31.2 μg/m³, the 24-hour average concentration of 50 μg/m³ is exceeded on 35 days. This is at the limit value for the 24-hour average. If correction for sea salt takes place, the number of exceedence days can be reduced by 6. In fact, the yearly average concentration is thus allowed to be so high that according to the above relationship, there could be 41 days with exceedence. The corresponding yearly average would be 32.3 μg/m³. This value is used to determine whether the Luvotool output results in exceedences of the daily norm for PM_{10}.

3.3.2 Luvotool in schematic form

Luvotool is the air module of EMPARA. The term "Luvo" is an acronym of the Dutch word for air pollution: *Lu*cht*v*er*o*ntreiniging.

The Luvotool model calculates the concentration map in accordance with the schematic shown in Figure 3.19. The model has two parts for calculating the contribution from traffic. For urban roads, the dispersion calculation is strongly parameterized. For non-urban roads, Luvotool assumes that a road section is a finite line source and the distribution is calculated with a Gaussian plume model.

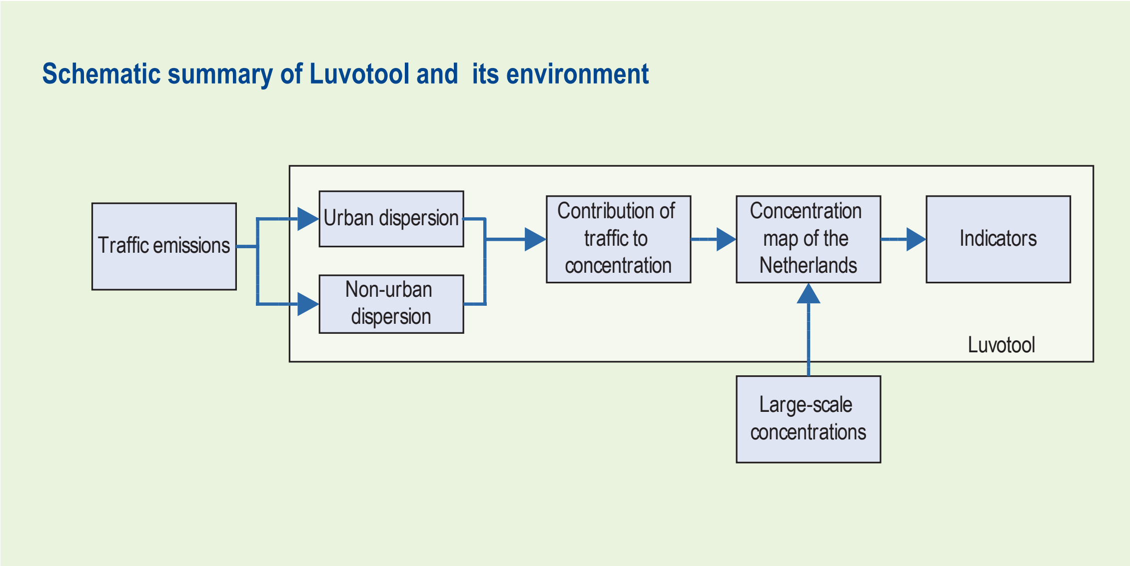

Figure 3.19 Calculation schematic in Luvotool. The text describes the steps that are distinguished in this figure.

3.3.3 Traffic emissions

Emission factors

Emission factors for road vehicles are determined by TNO [6]. For this purpose, since 2007 TNO has been using the VERSIT+ calculation model (Smit et al., 2006a+b). This model uses statistical modelling that is based on a large database with emission tests. In addition, the calculations are conducted with a much larger number of model classes than in previous versions. In VERSIT+, the traffic situations are described in more detail and are quantified by using trip patterns that provide an improved simulation of reality. The calculation models for heavy commercial vehicles were updated with recent measurement data from abroad and adapted to the latest insights. For use in the calculation models for air quality, the results from VERSIT+ are strongly aggregated to emission factors for a few vehicle and speed classes. This aggregation is conducted by TNO, while the PBL supplies the figures for the aggregation concerning traffic performance. For the vehicle classes passenger cars, medium-heavy vehicles, heavy vehicles and buses, this led to the emission factors for the five speed classes shown in Table 3.13.

Share directly emitted NO_2

Recently it has become clear, partly from international data, the average share of 5% directly emitted NO_2, which until recently was assumed, is too low. This led to an underestimation of the potential NO_2 concentrations in the vicinity of roads, especially for future years. The effect will increase the coming years because there is a direct connection with the renewal of the fleet. The emission factors that have been published from spring 2007, contain higher rates "directly emitted NO_2 for the different vehicle classes ", see Table 3.14.

6) Netherlands Organization for Applied Scientific Research – TNO, Division Science and Industry

Table 3.13 Speed classes as defined for CAR and the line source model.

Speed class	Definition
motorway, general	Typical motorway traffic, an average speed of about 65 km/h with about 0.2 stops per travelled kilometre on average.
non-urban road, general	Typical non-urban traffic, an average speed of about 60 km/h with about 0.2 stops per travelled kilometre on average.
urban traffic with less congestion	Urban traffic with a relatively large share of “free-flow” driving behaviour, an average speed between 30 and 45 km/h, an average of approximately 1.5 stops per travelled kilometre.
normal urban traffic	Typical urban traffic with a reasonable level of congestion, an average speed between 15 and 30 km/h, an average of approximately 2 stops per travelled kilometre.
stagnating urban traffic	Urban traffic with a high level of congestion, an average speed below 15 km/h, an average of approximately 10 stops per travelled kilometre.

Table 3.14 Share direct emitted NO_2 emissions for passenger traffic. For medium-and heavy traffic and buses the share is 0.07 for all years and speed classes.

	Urban road Stagnant	Urban road Normal	Urban road Moving	Secundary road	Motorway
2007	0.20	0.19	0.19	0.21	0.27
2010	0.24	0.23	0.23	0.28	0.31
2015	0.27	0.27	0.27	0.34	0.34
2020	0.27	0.27	0.28	0.31	0.31

Emission calculations in Luvotool

Traffic emissions per road section are calculated as follows:

$$E = Ef_l \cdot f_l \cdot I + Ef_{mz} \cdot f_{mz} \cdot I + Ef_z \cdot f_z \cdot I \cdot \frac{1000}{24 \cdot 3600}$$

E = emissions (µg/m/s)
Ef = emission factor (grams per kilometre).
f = fraction of traffic class.
I = traffic intensity (vehicles per 24-hour period).
indexes:
l = light vehicles
mz= medium-heavy vehicles.
z = heavy vehicles

3.3.4 Urban dispersion

Luvotool uses the CAR model (Calculation of Air pollution from Road traffic) to calculate the traffic contribution on urban streets. CAR has been used in the Netherlands since the 1980s. The model is calibrated to measurements from the National Air Quality Monitoring Network (*Landelijk Meetnet Luchtkwaliteit* - LML). In various sequential pieces of legislation, the government has prescribed this model as a reference. Recently, this model was established as Standard Calculation Method I (known by the Dutch abbreviation SRM I) in the Air Quality Evaluation Regulation, which went into force on 15 November 2007. The description of the calculation methods shown below was derived from that regulation.

Table 3.15 Road types as defined for CAR.

Road type	Definition
1	Road through open terrain, incidental buildings or trees within a radius of 100 m
2	Basis type, all roads other than types 1, 3a, 3b or 4
3a	Buildings on both sides of the road, distance between road axis and façade is less than 3 times the height of the buildings, but more than 1.5 times.
3b	Buildings on both sides of the road, distance between road axis and façade is less than 1.5 times the height of the buildings (street canyon)
4	Buildings on one side of the road, more or less continuous, distance between road axis and façade less than 3 times the height of the buildings

Input data for dispersion calculations with CAR

Road types
For calculating the concentration along the road, five road situations (=F road types) are distinguished. The road type depends on the height of the buildings along the road and the distance of the buildings to the road axis. Table 3.15 shows the defined road types. This data has been included for each road section in the urban roads database (Section 3.2.3).

Tree factor
The CAR model has the option of correcting for the effect that trees along a street can have on the dispersion of air pollution. This is done with a correction factor of 1 (no effect), 1.25 (moderate effect) or 1.5 (maximum effect). Luvotool uses a generic tree correction factor of 1.1 as an average for all urban streets; this is because data about trees along streets are not available on a national scale.

Meteorology
This section is largely derived from the Meteorology report in CAR II (Mooibroek and Wesseling, 2007). Wind direction and wind speeds are important parameters for the dispersion of air pollution. The CAR model does not take account of wind direction, but does include wind speed in the calculations. For this purpose, regional factors have been derived for six regions in the Netherlands, which are related to the annual average wind speed in the corresponding region, see Figure 3.20. In the CAR calculations, the concentration contribution is inversely related to the annual average wind speed. CAR is calibrated to a standard average wind speed of 5 m/s. The regional factor is defined as the ratio between this standard average wind speed and the annual average wind speed in the corresponding region.

The annual average wind speed per region is calculated using data from the meteorological stations of the Royal Dutch Meteorological Institute (KNMI) in De Bilt. The contribution of each station to that average is weighted proportionally according to the representativeness of that station within the region.

The region in which a road section is located is determined by the geographical coordinates of the road section (included in the roads database).

Calculating the dispersion

CAR calculates the concentration contribution at a receptor point by calculating the amount of dilution at that point. This takes place for road types 2, 3a, 3b and 4 with the formula:

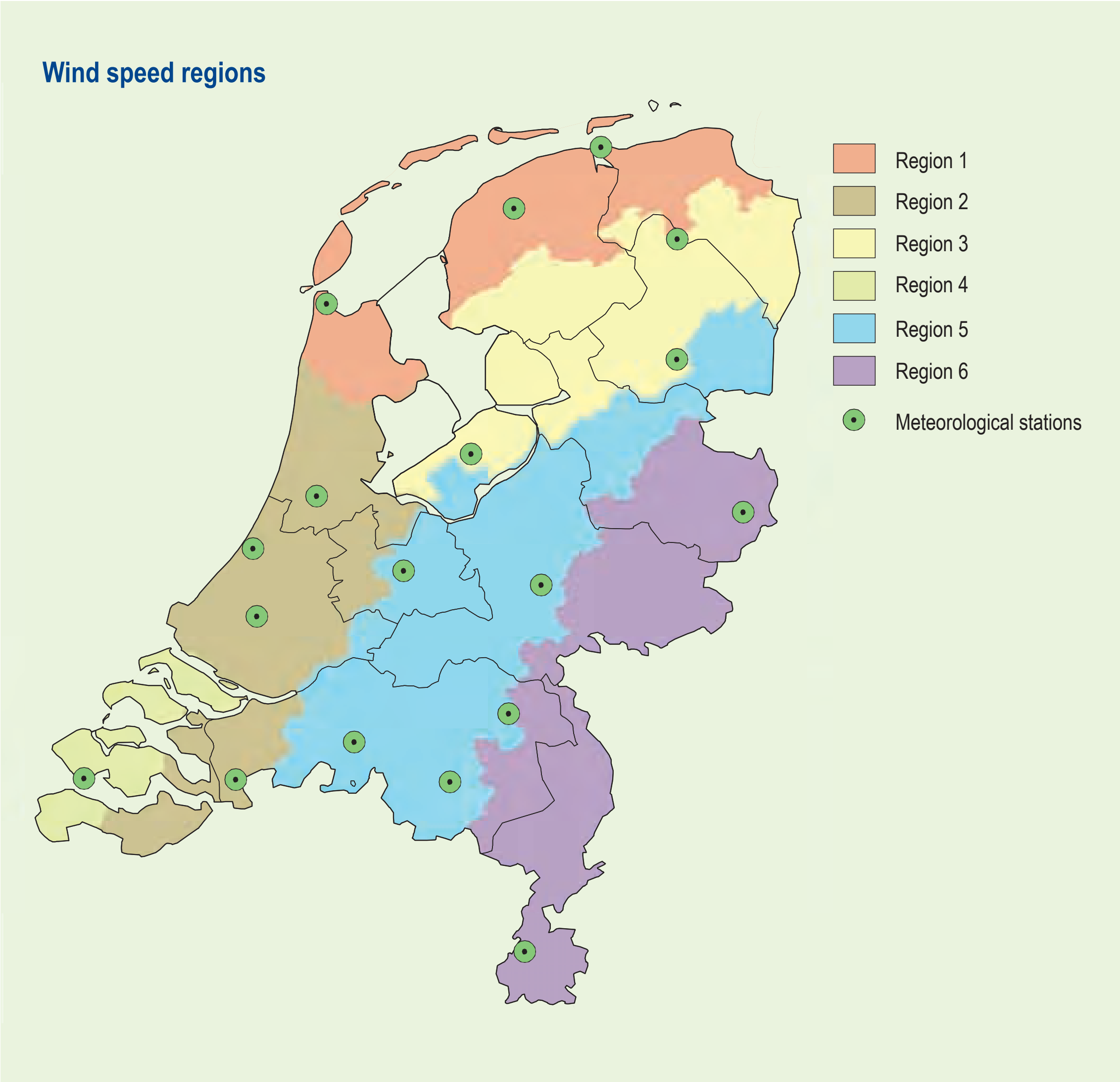

Figure 3.20 Wind speed regions for use in CAR and the location of the KNMI stations which provide the data.

$$\theta = a \cdot S^2 + b \cdot S + c$$

θ = dilution factor
S = distance to road axis
a,b,c = parameters, depending on the road type

For road type 1, the dilution factor is calculated with:

$$\theta = a \cdot S^{b\frac{S+e}{S}} \cdot (c \cdot S + d)$$

Table 3.16 contains the values of the parameters for the different road types.

Table 3.16 Road types as defined for CAR.

Parameter	Road type				
	1	2	3a	3b	4
a	0.725	$3.1 \cdot 10^{-4}$	$3.25 \cdot 10^{-4}$	$4.88 \cdot 10^{-4}$	$5.00 \cdot 10^{-4}$
b	-0.77	$-1.82 \cdot 10^{-2}$	$-2.05 \cdot 10^{-2}$	$-3.08 \cdot 10^{-2}$	$-3.16 \cdot 10^{-2}$
c	-0.0011	0.33	0.39	0.59	0.57
d	1.20	n.a.	n.a.	n.a.	n.a.
e	2.70	n.a.	n.a.	n.a.	n.a.

Contribution from traffic to the concentration

The yearly average concentration contribution of the traffic is calculated with the following formula:

$$C_{jgem} = F_{cor} \cdot E \cdot \theta \cdot F_b \cdot F_{reg}$$

C_{jgem} = yearly average concentration contribution
F_{cor} = correction factor, see below
E = emissions
θ = dilution factor
F_b = tree factor
F_{reg} = region factor (meteorology).

The correction factor (F_{cor} in the above formula) was included in 2007. After calculating with new and more reliable emission factors for road traffic (obtained with Versit+, see Section 3.3.3, it turned out that the CAR model structurally overestimated the traffic contribution with respect to the measurements. Additional research indicated that CAR had to be recalibrated. This led to the correction factor of 0.62. See also the textbox *Recalibration of the CAR model*

Conversion of NO_x to NO_2

The traffic contribution to the NO_2 concentration at the receptor point is calculated in terms of NO_x. This is followed by conversion to NO_2 as follows:

$$C_{jgem\_NO2} = F_{NO2} \cdot C_{jgem\_NOx} + \frac{B \cdot C_{achtergrond\_O3} \cdot C_{jgem\_NOx} \cdot (1 - F_{NO2})}{C_{jgem\_NOx} \cdot (1 - F_{NO2}) + K}$$

$C_{jgem—NO2}$ = yearly average concentration contribution NO_2 of the traffic
F_{NO2} = fraction of directly emitted NO_2
C_{jgem_NOx} = yearly average concentration contribution NO_x of the traffic
$C_{achtergrond\_O3}$ = background concentration of ozone
B, K = empirically ascertained parameters for converting NO to NO_2

Processing the urban traffic contribution

After the traffic contribution to the concentration at the location of the façade is calculated, these values are placed on a 25 × 25 m grid. This operation is required in order to combine the urban

Recalibration of the CAR model

(Wesseling and Sauter, 2007)

Research conducted by the RIVM has shown that the CAR II program systematically overestimated the traffic contributions. Recalibration of the model has greatly improved the agreement between the calculated and measured concentration levels. Local governments use the CAR II model (Calculation of Air pollution from Road traffic) to calculate air quality in traffic-loaded situations. Since the beginning of 2007, calculations have been conducted with new emission factors for road traffic which are significantly higher for various compounds than was previously the case. These new emission factors have led to significant differences between calculated and measured concentrations. On behalf of the Ministry of VROM, the RIVM conducted a new calibration of the CAR II model. Based on the traffic data supplied by municipalities, calculations were conducted with CAR II for the years 2003 through 2006 in the streets where measurement stations of the National Air Quality Monitoring Network of the RIVM are located. The results of the calculations for nitrogen oxides, carbon monoxide and particulate matter were then compared with measurements taken in the streets during the same period. The comparison showed that the calculated concentration contributions had to be multiplied by 0.62 to achieve good agreement with the measured values. Because the emission factors used in CAR II were also modified in 2007, the net calculated concentration levels in the streets will only change slightly with respect to past years. Following the recalibration, the calculated nitrogen dioxide and particulate matter concentrations are 0.4 and 0.7 μg/m³ (micrograms per cubic meter) higher on average than the measured concentrations. Without the recalibration, the calculated nitrogen oxide and particulate matter concentrations would be 5.7 en 2.4 μg/m³ higher on average than the measured concentrations.

concentration contribution with the non-urban contribution, which is calculated on a grid. The gridding takes place with a standard GIS operation.

3.3.5 Non-urban dispersion

The engine with which Luvotool calculates the traffic contribution to the concentration along non-urban roads is based on the VLW model (Prediction system for air quality on road sections - *Voorspellingssysteem Luchtkwaliteit Wegtracés*), which has been used for a number of years by the Directorate for Public Works and Water Management for reporting air quality along motorways. This model calculates dispersion according to the principle of a Gaussian plume model.

Recently, this VLW model was established as Standard Calculation Method II (known by the Dutch abbreviation SRM II) in the Air Quality Evaluation Regulations, which went into force on 15 November 2007. The description below is largely derived from these Regulations.

The model is suitable for calculating the dispersion of air pollution in situations without significant obstacles such as buildings and tall trees. The model takes account of the orientation of the road with respect to the wind direction. The description of SRM II includes the possibility of calculating with raised or lowered elevations of the road and with noise barriers. These possibilities have not been included in Luvotool.

Luvotool calculates the non-urban contribution from traffic on a grid of receptors within a strip along the road. The grid size and the width of the strip (calculation radius) can be adjusted. Recent experiences in practice have shown that a grid size of 25 m and a calculation radius of 1000 m provide a good balance between calculation time and the precision of the calculations.

Calculating the dispersion

The calculated emissions are linked to road sections from the roads database. Luvotool calculates the road segments as finite line sources; this differs from SRM II, which calculates a road segment as a series of point sources.

Figure 3.21 illustrates a number of definitions that are used in this section.

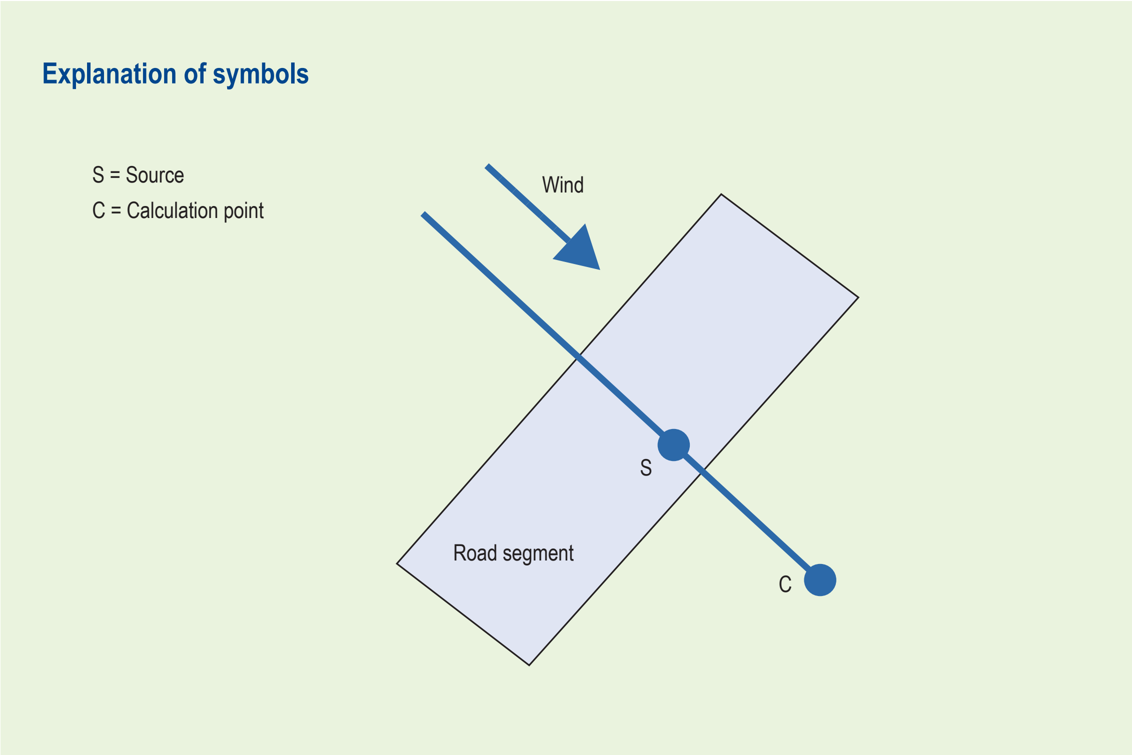

Figure 3.21 Meaning and position of symbols used.

An imaginary line BR is drawn between a source at position (B) within the road segment and a calculation point at location (R) inside or outside the road segment. On this line, the dispersion of the emission is assumed to be Gaussian. The direction of the line BR determines to which wind sector *i* the contribution *Cb,i* of source B to the concentration at calculation point R belongs. This contribution is calculated using the following plume formula:

$$C_{b,i} = \frac{E \cdot d_w}{\sqrt{2\pi} \cdot \sigma_z \cdot C \cdot u} \cdot \frac{1}{\pi R_B / n} \cdot \exp \frac{-(z-h)^2}{2\sigma_z{}^2}$$

E: the emissions per length unit [µg.m-1.s-1];
dw: the length of a road segment [m];
RB: The distance from the source (B) to the calculation point (R), i.e. the calculation distance [m];
σz: the vertical dispersion coefficient [m];
z: the elevation of the calculation point [m];
C: a roughness-dependent correction factor [-];
u: the wind speed [m/s];
n: the number of wind direction sectors (12);
h: source elevation.

For each calculation point, this calculation is performed for all source positions. Luvotool here differs from the standaard method and describes the section as a finite line source, and calculates per windsector the integral of the line source on the angle viewed from the calculation point. The result of the integral is similar to the above formula: a function T, depending on the viewing angle, replaces the factor d_w / R_b.

The emission (E), the correction factor (C), the vertical dispersion coefficient (σz) and the wind speed (u) are explained in further detail below.

Correction factor (C)
The correction factor C corrects for a number of effects and is calculated with the following formula:

$C = C_{wind} * C_{meteo} * C_{24h}$

The correction (C_{wind}) for the velocity profile of the wind is calculated from the height of the plume (z_p) and the average stability of the atmosphere. For z_p, it is assumed that this is equal to 75% of the vertical plume dispersion: $z_p = 0.75\ \sigma_z$.

$$C_{wind} = \frac{\ln \frac{z_p}{z_0} - \psi \frac{z_p}{L} + \psi \frac{z_0}{L}}{\ln \frac{z_{10}}{z_0} - \psi \frac{z_{10}}{L} + \psi \frac{z_0}{L}}$$

where the correction ψ is given by:

$$\psi \frac{z}{L} = -17\ 1 - e^{-0.29\frac{z}{L}}$$

The function ψ is dependent on the atmospheric stability through the value of the Monin-Obukhov length (L) that is used. The factor C_{meteo} corrects for the effective conversion of the roughness length at the location of Schiphol or Eindhoven to the roughness length used for the calculation. The parameters are:

Roughness length	L	C_{meteo}, Schiphol	C_{meteo}, Eindhoven
0.03	60	0.7000	0.7000 * 0.95
0.10	60	0.7050	0.7050 * 0.95
0.30	100	0.6525	0.6525 * 0.95
1.00	400	0.7400	0.7400 * 0.95

Luvotool works with continuous values for the roughness length. For L, the following relationship is therefore derived that satisfies the discrete values in the above table:

$L = -501.8z_0^3 + 956.53z_0^2 - 117.37z_0 + 62.674$; C_{meteo} does not show an unequivocal relationship with the roughness length and is therefore held constant in Luvotool: $C_{meteo} = 0.7$.

The correction factor C_{meteo} for the progression of the meteorological conditions during the 24-hour period corrects for the fact that a calculation over 24 individual hours, each with their own emission and meteorological conditions, provides a different result than a single average calculation for those 24 hours. For example, traffic emissions are highest during the day, while the wind speed is higher on average during the day than at night. The value of the correction is constant in all cases: $C_{24h} = 1.15$

Vertical dispersion coefficient (σ_z)
The vertical dispersion coefficients are fitted to the results of calculations with the New National Model (*Nieuw Nationaal Model*). SRM II calculates the vertical dispersion as follows:

$$\sigma_z = \frac{a \cdot R_B{}^b}{1 + 0{,}5 \cdot \left(1 - e^{-(R_B/2800)^2}\right)} + \sigma_{z0}$$

R_B = distance between source and receptor point.
$\sigma_{z,o}$ = fictive vertical displacement of the source due to the initial dispersion
a and b = parameters dependent on the roughness length.

Table 3.17 Value of the parameters a and b in SRM II

Roughness length	a	b
0.03	0.2221	0.6574
0.10	0.2745	0.6688
0.30	0.3613	0.6680
1.00	0.7058	0.6207

Luvotool works with continuous values for z_0. In Luvotool, the parameters a and b from Table 3.17 have therefore been replaced with:

$a = 0.02126(\ln z_0)^3+0.1645(\ln z_0)^2+0.45337(\ln z_0)+0.7058$
$b = -0.00351(\ln z_0)^3-0.02906(\ln z_0)^2-0.06919(\ln z_0)+0.6207$

The term $\sigma_{z,o}$ in the equation above indicates the initial dispersion. Luvotool differs from that equation because it calculates the initial dispersion by fictively placing the source at a larger distance from the receptor. This makes σ_z become a function of $(x + x_0)$ instead of (x).

This leads to the following equation for Luvotool:

$$\sigma_z = \frac{a \cdot (R_B + x_0)^b}{1 + 0{,}5 \cdot \left(1 - e^{-((R_B + x_0)/2800)^2}\right)}$$

x_0 is the fictional horizontal distance needed to start with a certain plume height at the spot of the source. The distance x_0 follows through iteration of a simplified model for the dispersion parameter σ_z (x), valid for the first tens of meters:

$$\sigma_{z0} = 0{,}2 x_0{}^{0{,}76} \cdot (10 z_0)^{0{,}53 x_0^{-0{,}22}}$$

The initial value for the vertical dispersion $\sigma_{z,o}$ depends on the type of surroundings:
– outside the built-up area, the road is not a motorway: $\sigma_{z,o}$ = 2.5 [m];
– outside the built-up area, the road is a motorway: $\sigma_{z,o}$ = 3 [m].

A practical problem for Luvotool is that the model can no longer make a distinction between motorways and other non-urban roads after emissions have been calculated. The initial value for the vertical dispersion has therefore been set to 3 m for all non-urban roads. This results in a small underestimation of the traffic contribution from non-urban roads that are not motorways (approximately 10%).

SRM II has the possibility of adapting $\sigma_{z,o}$ to the raised or lowered elevation of the road or the presence of barriers. Luvotool does not yet have these options. This is not a technical problem; national databases with this data are not available.

Until recently, the calculation for σ_z as used in Luvotool was used in most air-quality models. However, the differences in the approach to the initial dispersion also lead to differences in results. For the next version of Luvotool, it has therefore been decided to adopt the approach used in SRM II.

Determining the yearly average concentration
The annual average wind speed is shown in a wind rose comprising 36 sectors of 10°; for each sector j, the harmonic mean wind speed u_j and the frequency of occurrence f_j are given. Wind roses are determined for 5 regions based on hourly observed values at a KNMI station representative for the region:

Region	KNMI station
1	270 Leeuwarden
2/4	240 Schiphol
3	280 Eelde
5	370 Eindhoven
6	375 Volkel

The division into regions is in accordance with that of the CAR model (figure 3.20) Wind roses are available for calendar years. For scenario calculations (future years), a multi-year wind rose is used which is based on all hourly values during the period 1981-2000.

The contribution from traffic at a receptor point is calculated by placing the wind rose on the receptor point. For each wind sector, the contribution from a road is determined using the Gaussian plume formula, where the emission is only taken into consideration to the extent that it lies within the corresponding sector. The wind speed u is equal to the harmonic mean wind speed from the wind rose; in the correction factor C, this wind speed is corrected as shown above. In this correction, it is assumed, among other things, that the wind speed at ¾ of the plume height is representative for the dispersion.

Because the plume usually comprises multiple wind sectors, for each wind direction, 12 adjacent sectors are also taken into account. The total contribution C_j per wind direction j is the weighted sum of concentration contributions per sector $c(i)$:

$$C_j = \sum_{k=-6}^{6} w(k) \cdot c(j+k)$$

with weighting factor $w(k)$:

$$w(k) = 8/36 \cdot e^{-\frac{1}{2}(20 \cdot k/36)^2}$$

Differences in wind roses between SRM II and Luvotool

As defined for SRM II, the wind rose comprises 12 wind sectors, subdivided into three wind speed classes. For each of these 36 classes, depending on the meteorological year, the frequency of occurrence is given. There are 2 meteorological regions.

In Luvotool, a wind rose with 36 wind sectors is used. For each sector, the harmonic mean wind speed and the frequency of occurrence are given. In Luvotool, 5 meteorological regions are defined.

Finally, the yearly average concentration at the receptor point is calculated from concentrations per wind direction and the frequencies of occurrence from the wind rose:

$$C_{average} = \sum_{j=0}^{36} f_j \cdot C_j$$

3.3.6 Contribution from traffic to the concentration

The output of the contribution from traffic on motorways and non-urban roads comprises a grid with 25x25 m cells (up to 1000 m from the roads), for which the contribution from traffic has been calculated . The output of the urban calculation (calculated with CAR) comprises the contribution from traffic to the concentration at the façade. This contribution is placed on a grid with 25x25m cells. For each cell on the grid, Luvotool sums the various contributions.

Adding two NO_2 sources (traffic or otherwise)
The NO_2 contribution of two sources (traffic or otherwise) cannot simply be added together due to the photochemical reaction that takes place between NO_2, NO and O_3. For Luvotool, a calculation method has been developed to add two such contributions.

NO_x-NO_2 conversion
The addition method is based on the NO_x-NO_2 conversion from the CAR model. In this method, a fraction of the NOx contribution is emitted directly as NO_2. The remaining fraction consists of NO and is partially photochemically converted into NO_2 depending on the available O_3 background concentration and two photochemical parameters. The NO_2 contribution of the source is the sum of the direct and indirect (converted) NO_2 contributions:

$$vNO_2 = fNO_2 \cdot vNO_x + (1 - fNO_2) \cdot \beta \cdot Ca\_O_3 \cdot vNO_x / (vNO_x + K) \qquad (1)$$

vNO_2 = NO_2 contribution of a source
vNO_x = emitted NO_x
fNO_2 = fraction of directly emitted NO_2
$1-fNO_2$ = fraction of NO
Ca_O_3 = background concentration of ozone
K and β= photochemical parameters: K =100, β = 0,6.

Method for adding source contributions.
The method assumes that two known NO_2 contributions can be reverse calculated to NO_x contributions, added as NO_x and then converted to a single NO_2 contribution. The calculation step from vNO_2 back to vNO_x is conducted by rewriting the equation above as a 2^{nd} degree equation with the following solutions for vNO_x:

$$vNO_x = \frac{-B \pm \sqrt{B^2 + 4 fNO_2 \cdot vNO_2 \cdot K}}{2 \cdot fNO_2}$$

where

$B = fNO_2 \cdot K + (1 - fNO_2) \cdot \beta \cdot Ca\_O_3 - vNO_2$

The minus in ± does not provide a real number solution. Generally speaking, all parameters are known, except for the fraction of directly emitted NO_2 (fNO_2). Until recently, it was assumed that this fraction was relatively constant and had an average value of approximately 5% for the national fleet of vehicles. However, new insights have shown that this fraction can be significantly higher for passenger cars (see Section 3.3.3). This means that for every situation where two or more NO_x source contributions must be added together, the factions of directly emitted NO_2 must be known, or otherwise the proportions of light and heavy vehicles. For Luvotool, a solution to this problem has not yet been found and the conversion is still performed on the basis of 5% directly emitted NO_2. This conversion does introduce an error, but it is limited because both the conversion to NO_x and the conversion back to NO_2 take place with the same value of 5%.

3.3.7 Large-scale concentration

The total of contributions from the local source plus the large-scale concentration determine the final air quality. The *large-scale concentration* is defined as the concentration that is calculated with a generic method on a scale of 1x1 km2 and that is based on all emission sources in the Netherlands and abroad. For model calculations of local air quality, such as near a road, the large-scale concentration is used in a traffic model to approximate the background concentration. The local air quality can then be defined as the sum of the calculated local contribution of the source plus the large-scale concentration.

The map of the Netherlands with the average concentrations comprises a grid with cells of 1x1 km². This grid is based on the OPS model (see text box), a database with emissions from the Netherlands and abroad and measurement results from the National Air Quality Monitoring Network. The OPS dispersion model is specifically suited for calculating contributions of emission sources to the yearly average concentration. In this model, the distance between the source and receptor can be very large. The emission databases are based on national total emissions for a number of separate source categories (such as traffic, energy and various industries) and a spatial allocation for these source categories.

In the case of NO_2, the NO_x fields calculated with the OPS model are used. The NO_x fields are converted to NO_2 and O_3 fields. For each grid cell, the concentrations of NO_2 and O_3 are derived from the NO_x concentration with the aid of an empirical relationship. This relationship is based on the yearly average concentrations of NO_x, NO_2 and O_3 from the National Air Quality Monitoring Network that were observed during the period 1991-2000.

The calculated grid (1x1 km²) is calibrated to measurement results of regional and city background stations (except for PM_{10}) from the National Air Quality Monitoring Network. There are several essential differences between the large-scale concentrations from historical years and future years. The most important differences are the following:

The OPS model

OPS is a model that simulates the atmospheric process sequence of emission, dispersion, transport, chemical conversion and, finally, deposition. The model is set up as a universal framework supporting the modelling of a wide variety of pollutants including particulate matter, but the main purpose is to calculate the deposition of acidifying compounds in the Netherlands at a high spatial resolution. Previous versions of the model have been used since 1989 for all the atmospheric transport and deposition calculations in the *State of the Environment* reports and *Environmental Outlook* studies in the Netherlands.

An extensive model validation exercise was carried out using observations from the National Air Quality Monitoring Network from the past 20 years. Good agreement was found with SO_x and NOy species in the spatial patterns and the trends over the past 10 years. The NH_x species was an exception; these were usually underestimated by about 25%. This discrepancy is referred to as the 'ammonia gap'. (Jaarsveld, 2004).

– model calculations for future years are always conducted with multi-year averaged meteorology. If emissions are kept constant, the variations in meteorological conditions that occur from year to year lead to fluctuations in the concentrations of approximately 10%. If a comparison is made with the present, defined as the basis year, then the basis year must also be calculated with multi-year averaged meteorology. This is necessary in order to make the effects of emission changes visible in the concentrations while excluding the effects of meteorological fluctuations.
– uncertainties in the final results are also determined by uncertainties in the assumed economic, societal and technical developments, in addition to the uncertainties in emission data and the dispersion model. For a future year, it is logical that no measurement results are available.

Large-scale concentrations for future years are based entirely on model calculations. In these calculations, the emissions are derived from scenario studies. At a higher level, assumptions are also made about economic developments (Central Planning Office). Emissions are estimated based on the predicted developments in human activity and in national and/or European policy. The results of calculations with the OPS model are post-processed in a comparable fashion as with the GCN of an actual year (NO_x-NO_2 conversion), but of course they are not fitted to the results from the measurement network. For PM_{10}, the large-scale concentrations in scenarios are corrected for the systematic differences between measurements and model calculations (due to unknown sources and natural contributions).

For practical reasons, only a single concentration map (per substance, per year) is delivered which is available for all model applications and is based on contributions of all known sources in the Netherlands and abroad. However, this may result in data duplications if the local effect of an existing source is calculated separately and then added to the large-scale concentration. The contribution of the source to the large-scale concentration is relatively low and in many cases, such as urban roads, is negligible. The duplication does become a problem if the contribution to the large-scale concentration, on a 5x5 km² scale, is significant. But this concerns only very strong emission sources, such as busy motorways and large industrial installations.

Correction for double counting emissions
(Velders et al., 2008)

The large-scale concentrations map represents the concentrations in The Netherlands on a 1 x 1 kilometer grid, coused by emissions by all domestic sources and the contribution of foreign sources.

Double counting occurs in cases where the contribution of an emission source to the concentration is calculated by a dedicated model and is added to the large-scale concentrations map. For small sources e.g. local roads, the extent of the double counting is negligible. However in case of major sources the extent of double counting requires a correction. This is the case in adding the contributions of national highways.

The correction for double counting in case of road traffic is estimated with a map of motorway emissions and dillution factors. Given one unit of emission from a grid cell the dillution factor represents the concentration contribution in that cell and 48 surrounding cells. The correction, estimated to within about 3.5 kilometers of a motorway, is calculated as follows:

$$C_{corr}(2007) = C_{lrg-scale}(2007) - \frac{e_s(2007)}{e_s(2005)} \cdot E_s(2005) \cdot f \cdot DF$$

C_{corr}	= concentrations map corrected for the double counting of the emissions from motorways
$C_{lrg\text{-}scale}$	= concentrations map on a 1x1 km-resolution
E_s	= map with emissions from motorways in the reference year 2005 (in kg km^{-2})
es	= total emissions from motorways
f	= conversion factor from kg km^{-2} to g s^{-1}
DF	= map with dillution factors (in ppb s g^{-1}), calculated with current or with long-term meteorological data.

The correction for double counting is calculated seperately for each vehicle category and then combined. The correction for NO_2 and O_3 is derived form the corrected NO_x using an empirical relationship.

3.3.8 Concentrations map of the Netherlands

Luvotool adds the map with traffic contributions (25x25 meter grids) to the large-scale concentrations map. The map with corrections for double counting is subtracted from the total. The result is a map of The Netherlands with total concentrations. From this map various indicators are derived which are the basis for policy analysis.

3.3.9 Calculating indicators

Indicator 'kilometres of road length with norm exceedances'

Urban roads
For urban roads, the concentration at the façade is calculated and linked to the corresponding road section. This concentration is used to test for exceedances [7)]. The lengths of all road sections are known; for urban roads it is therefore possible to determine the kilometres of road length with norm exceedences by using a simple GIS analysis.

7) This is not entirely correct for two reasons. Firstly, this method does not include the exceedences along road sections without buildings, and secondly this does not comply with the criteria for the distance to the receptor point as defined in the legislation (10 m distance from the road edge or the distance to the façade, if this is closer). This is a point for improvement for the next version of Luvotool.

Motorways
For motorways, the exceedance is determined at 20 m from the middle of the road. On average, this distance is equivalent to 10 m from the edge of the road, as required by the legislation. Due to the calculation method used for motorways, after calculation there is no id-link between the road section and concentration. This requires an indirect approach. By means of a GIS operation, all grid cells are selected from the total map for which the midpoints lie at 18 to 22 m from the road axis. The total length of the motorways is then divided by the selected grid cells. In this way, every grid cell represents a specific road length. It is then determined which percentage of these grid cells exceeds the norm. This percentage is multiplied by the road length per point, which gives the length of motorways with exceedences of the norm.

For secondary roads and other non-urban roads, the same method should be used as for motorways. An analysis of Luvotool results for the year 2006 showed that norm exceedences occur on less than 0.5% of these roads. Due to the magnitude of the analysis and the low level of exceedences on these roads, it was decided not to conduct this analysis.

Indicator 'exposure'
For determining the exposure, besides the national map with the total concentration, a database that includes all houses and the number of residents per house is also required. The database that is used for this purpose is the housing and population database, as available from the PBL 'data portal'. With this data exposure can be calculated at the house level. For each house, the exposure to the component that was calculated beforehand is determined. As a result, the number of individuals and locations with a concentration that exceeds the norm can be specified.

Effects of measures
The effects of European and national measures to improve air quality primarily come to effect in the large-scale concentration maps. Luvotool adds the contribution of the traffic to the large-scale map and shows the change in kilometers of roads that exceed the limit values. Luvotool adds additional information in case the measures effect the emissions of the traffic.

Luvotool is not designed for calculating the effects of specific local measures. The effects of regional and local measures can be roughly estimated and are taken into account for the 'local situation' in general. Only very general statements referring to the national level can be made about this topic.

3.3.10 Validation and uncertainties

The aim of Luvotool is the following: to make a national survey of the magnitude and type of air quality hotspots and to indicate the areas in which the hotspots are located and the influence of national and international policy measures on them. To achieve this aim, exact location-specific analyses are not required, but a correct average diagnosis must be made across a larger area (for example the urban region of the Randstad, other large cities). The quality of the input is compatible with this analysis requirement. For the purposes of the validation, the methods and the input data of Luvotool were analyzed regarding systematic deviations with respect to measurements and other models.

Calibration of the calculation method for urban situations (CAR)
The CAR model has been calibrated by the RIVM using measurements from the National Air Quality Monitoring Network (Wesseling and Sauter, 2007). For this purpose, measurement

Table 3.18 Deviation of the CAR model with respect to street stations of the National Air Quality Monitoring Network.

	NO_2 (µg/m³)	PM_{10} (µg/m³)
Average	0.4	0.7
Standard deviation	4	6

results were used from 14 measurement stations in busy streets for the years 2003-2006. These stations were located in 11 large cities in the Netherlands. The CAR calculations have been conducted with locally obtained input of road and traffic characteristics that are linked to the locations. The conclusions from this calibration are that the NO_2 and PM_{10} concentrations are on average not significant (0.4 and 0.7 µg/m³) higher than the measurements, respectively, with standard deviations of 4 µg/m³ (NO_2) and 6 µg/m³ (PM_{10}). The calculated yearly average values cartainly fall within the uncertainty margins of 30% (NO_2) and 50% (PM_{10}) that are prescribed in the legislation for these components.

A comparison of the Luvotool results with those from the official CAR model shows that the CAR algorithm has been implemented correctly in Luvotool.

Calibration of the calculation method for non-urban situations (dispersion model)
This part of Luvotool is based on a model description of the TNO Traffic Model (TNO-VM) from 1993 (TNO, 1993a and 1993b). The most important methodological differences between the Luvotool Dispersion Model (LDM) and TNO-VM are:

1. the description of a road section as a source (a single finite line source instead of a series of point sources),
2. the calculations in LDM are conducted with a roughness length (z_0) between 0.03 en 1.2 m, while TNO-VM uses four categories (0.03, 0.1, 0.3 en 1.0 m),
3. partition of The Netherlands into five meteorological regions instead of two,
4. the wind rose that is used (for each wind sector, a single harmonic mean wind speed instead of three wind speed classes); this wind rose is corrected using weighting factors for the effect of the 24-hour progression of traffic emissions,
5. for the conversion of the contribution of NO_x from traffic into NO_2, LDM uses a single yearly average ozone concentration, while in TNO-VM the available ozone is made dependent on the wind direction.

Points 1-3 are an extention of the model and are regarded as a methodological improvement. Point 4-5 form a simplification of the calculation method without adverse effects to the outcome of the calculations. Both approaches (for wind speed an – frequency as well as NO_x-NO_2 conversion) are subject to discussion.

During the development, the calculation results from LDM were in good agreement with those from TNO-VM (Van Velze, 2004). Differences in outcome are less than 2% for inert substances (such as NO_x and PM_{10}). The difference between a finite line source and a series of point sources is noticeable at a short distance from the source (< 20 m from the road axis), but is not a problem because for motorways these receptor points lie above the road surface and can be excluded. Differences in meteo-wind roses are expressed in the results, which is expected. In LDM, the correction for the 24-hour progression is made by using a factor derived from calculations with TNO-VM. In case of NO_2 the difference between LDM and TNO-VM is 4% at the most for the total concentration and relatively higher for the contribition of the traffic alone, however less than 8%. These differences can be attributed to the use of the ozone-wind rose in TNO-VM, which has an effect of dissymmetry to the concentrations on both sides of the road. In LDM, the effect of the ozone wind rose on the NOx-NO_2 conversion is partly taken into account by calibrating LDM to TNO-VM. The result of this calibration is that one of the two photochemical parameters (ß) is made distance-dependent. Nevertheless, the cross-section profile [?]of concentrations is more strongly dependent on the road orientation in TNO-VM, which has wind roses for both wind speed/frequency and ozone, than in LDM, which has a wind rose only for wind speed/ frequency. An ozone wind rose was not modelled in LDM for technical reasons (the calculation time is approximately 36 times longer), and due to the suspicion that the ozone wind rose in an urban environment is strongly affected by the source areas in the vicinity.

The Calculation and Measurement Regulations (known by the Dutch abbreviation MRV) were published in 2006; they defined two standard calculation methods, of which the Standard Calculation Method 2 (SRM2 or SRM II) is intended for motorways. SRM2 is based on the Plume motorway model of TNO (TNO-PS). TNO-PS was in turn based on TNO-VM; during the preparations for SRM2, a number of improvements were made in the model. These modifications to TNO-PS, which were included in SRM2, comprise the following:

- calculation of vertical dispersion ($\sigma_z(x)$) from the roughness length (z_0),
- dealing with the initial dispersion ($\sigma_{z,0}$) caused by turbulence behind vehicles,
- the addition of the roughness-dependent correction factor (C), based on three sub-factors: correction for the wind speed profile (C_{wind}), correction for the effective conversion of roughness length of the measurement station to the calculation situation (C_{meteo}) and correction for the 24-hour progression of traffic emissions (C_{etmaal}).

The current version of LDM has been partly adapted in accordance with the changes in SRM2 in relation to TNO-VM. Two parts not (yet) in accordance with SRM2 are:

1. the initial dispersion ($\sigma_{z,0}$), which is still dealt with according to the old method. In the newly prescribed method, $\sigma_{z,0}$ is added to $\sigma_z(x)$; in the old method, the "starting point" of the plume is moved from the source to a virtual point in such a way that the height of the plume at the location of the source is equivalent to the initial dispersion.
2. the roughness-dependent correction factor: from this part the components C_{wind} and C_{etmaal} have been included in LDM, but C_{meteo} has not. The dependency of C_{meteo} on z_0 shows an irregular behaviour and a smooth function could not be derived. Instead, the average value of C_{meteo} is calculated in the current version LDM. A better method is lineair interpolation between the four standard values of z_0 in SRM2.

These two parts will be implemented in the next version of LDM.

The various implementations of SRM2 (TNO-PS, VLW, KEMA-Stacks, RIVM-SRM2) do not always generate the exact same results. Calculations with the current version have been compared with results from the RIVM implementation of SRM2. This comparison of results (Van Velze, 2008) showed that LDM deviates systematically, and in most cases calculates higher concentrations. At distances of 25 m or more, the results of LDM for inert substances are at the most 30% higher, and 8% for NO_2. In a stand-alone version of LDM, the changes in SRM2 that have not yet been included (dealing with initial dispersion and C_{meteo}) have been included and the effects on the results have been compared. This comparison showed that the results were more in agreement after modification. Following the two modifications, the results of LDM for inert substances are about 10% higher for NO_2 they are about 1% higher. The deviations remaining after

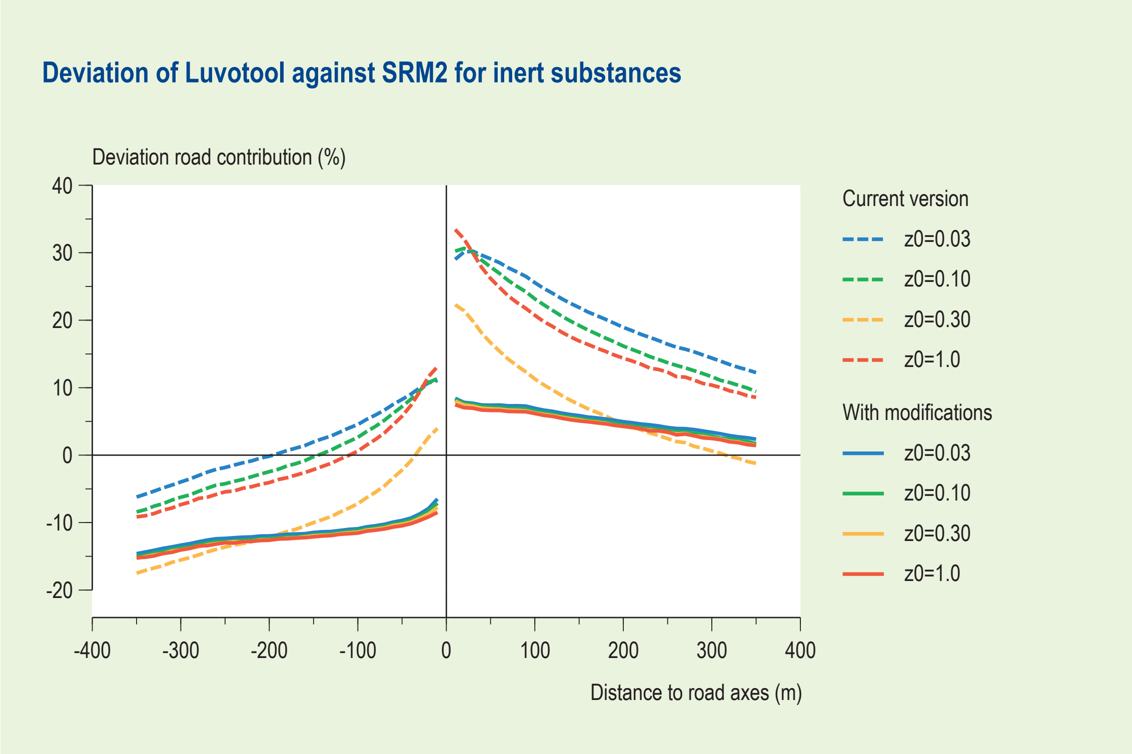

Figure 3.22 Percentage deviation of the contribution of traffic for inert substances, calculated with LDM in its current version and with the modified version ($\sigma_{z,0}$ and C_{meteo}), calculated with a multi-year wind rose, for various values of z_0.

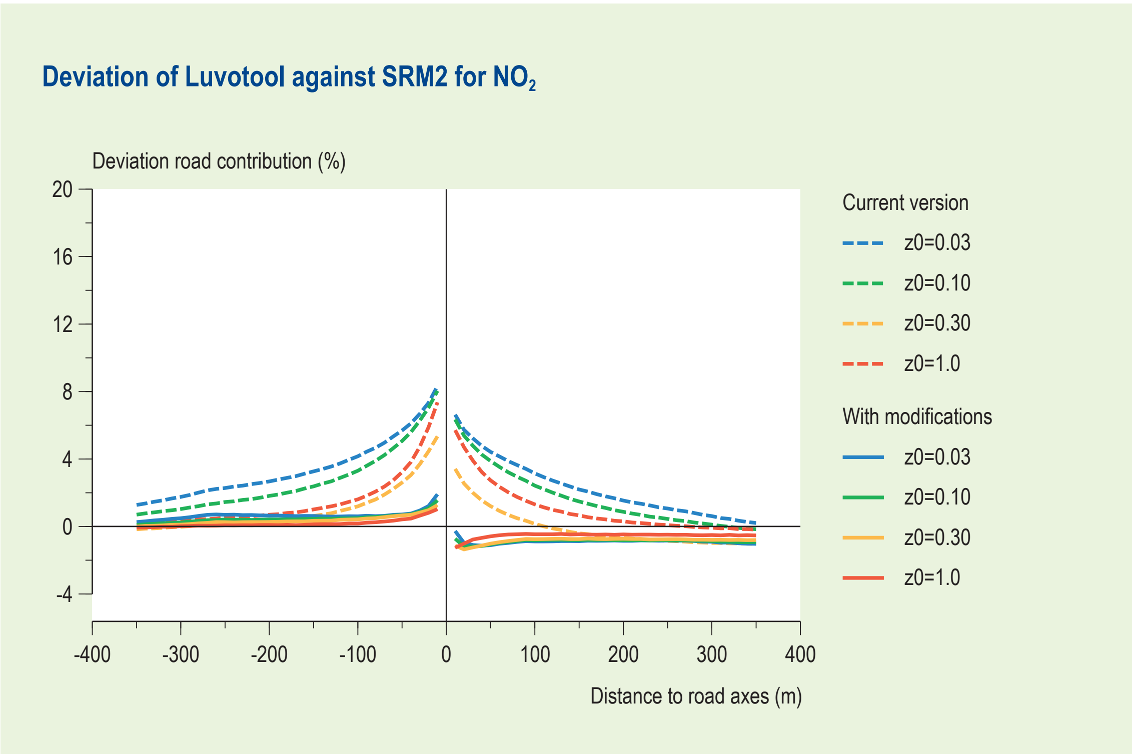

Figure 3.23 Percentage deviations of the NO_2 contribution calculated with LDM in its current version and with the modified version ($\sigma_{z,0}$ and C_{meteo}), calculated with a multi-year wind rose, for various values of z_0.

modification have been attributed to differences in the meteorological wind roses used, and for NO_2, to the ozone wind rose.

Validation of input data

GCN
The Generic Concentration Maps of the Netherlands (GCN) are calculated with the OPS model, see Chapter 4 (audit report). The uncertainty in the background concentration is estimated at 15% (1σ) for both NO_2 and PM_{10} (Velders et al., 2008).

Luvotool-CAR: urban data
To calculate the contribution to air pollution from traffic on urban roads, Luvotool uses the CAR model with the corresponding data about traffic and roads, which will be referred to here as the CARdata. Much of the CARdata for Luvotool was calculated in a generic fashion (see Chapter 4 of the audit report). The present section explores what this means for the reliability of the Luvotool results. This will be done by comparing the results from Luvotool with those from the Traffic Environmental Maps (*Verkeers MilieuKaarten* - VMKs) of a number of large municipalities. In both cases, the contribution from traffic will be calculated with CAR. The comparison therefore concerns only the input data. The point of departure is that the input data of the VMKs is based on good knowledge about the local situation, and therefore must be better than the generically constructed input. It is difficult to determine the absolute quality of the VMKs, and this quality is not known at present. In this comparison, however, the VMK is used as the standard, and the analysis indicates to what extent Luvotool deviates from this standard.

Adaption tree factor
The tree factor indicates by a factor 1, 1.25 or 1.50, to what extend dillution of air pollution is hindered by trees in the street. As there is no database with nationwide data on this, an average tree factor of 1.11 is derived from VMKs of 40 municipalities. This value was included in Luvotool.

Calculation distance
The calculation distance is the distance from recepot point to the middle of the road. Luvotool uses two kinds of receptor points: kerbside, with a standard distance of 7 m, and façade distance. For Luvotool, the façade distance is determined by combining various geographical databases. With the exception of complex situations or streets with large variations in façade distance, this method is very precise because these geographical databases are extremely accurate. Random samples have shown that the deviations in the façade distance in the VMKs that were used (which are now out-of-date) are larger than those in Luvotool in many cases. One municipality applied a standard calculation distance of 9 m to all situations. The differences can partly be attributed to the lack of regulations concerning the calculation distance. However, the new Air Quality Act, which went into force on 15 November 2007, does contain regulations about the calculation distance. For NO_2 and PM_{10}, this legislation prescribes maximum calculation distances of 5 and 10 m, respectively, measured from the edge of the road, or to the façade if this is closer. During the course of 2008, new EU standards will go into force, which prescribe a calculation distance of 10 m for both components. Luvotool has not yet been modified in accordance with these new Dutch and European standards.

Validation of CAR input data with measurements
The input data of the CAR module has been validated with data from LML street stations from the same set as that which was used for the calibration of the CAR model described above. A number of stations were eliminated because Luvotool had not calculated a contribution from traffic at those locations. In most cases, this was because the façade distance at those locations was greater than 30 m (in which cases no contribution from traffic is calculated). At two of the stations the contribution from traffic was missing because the road on which these stations are located did not appear in the roads database. These stations were eliminated because they confuse the comparison with measurements. However, it is important to obtain insight into how often these types of situations occur, because they do affect the total result of Luvotool.

The comparison with measurements was conducted for 2006 (the values from the municipalities – see Wesseling and Sauter (2007) – were corrected for the elevation difference between measurements and calculations (approximately 3 m and 1.5 m, respectively). For an analysis of the Luvotool results, a number of Luvotool calculations were simulated with a stand-alone CAR model. Table 3.19 shows the various steps that were taken during the comparison.

The measurement and calculation results are shown in columns 3 and 4. These results are not directly comparable because Luvotool calculates at façade distance, while the measurement points are located at other distances.

Column 5 shows the results of calculations with location-specific input (instead of the generic input of Luvotool) at the exact distance of the measurement points. RIVM determined this data in cooperation with the municipalities in order to calibrate the CAR model (see above). These are the best available calculations to compare with the measurements.

Column 6 contains the same calculations as column 5, but at façade distance, as in Luvotool. These are the best available calculations of the concentrations at façade distance. The deviation of Luvotool from these calculations provides an indication of the deviation caused by the use of generic input versus local-specific input.

Column 7 shows the results of calculations with the Luvotool input, with the calculation distance to the measurement points instead of the to the façade. This is the best possibility to compare the Luvotool calculations with the measurements.

Column 8 contains partial results from the previously mentioned calibration of the CAR model. The result for this data subset is comparable with the calibration.

Column 9 shows the deviation resulting from the use of generic input: an average underestimation of 3.5 μg/m³, with a standard deviation of almost 11 μg/m³.

Column 10 shows the final result of the validation: in urban situations there is an underestimation of 3 μg/m³, with a standard deviation of almost 10 μg/m³.

Conclusion about urban calculations
On average, Luvotool underestimates by approximately 3 μg/m³ with respect to measurements. However, the relative standard deviation of the differences is large (standard deviation 10 μg/m³); consequently, considering the number of comparisons made, the deviation is not significant. However, there is a high probability that Luvotool underestimates by several micrograms rather

Table 3.19 Analysis of differences between urban measurements and calculations, NO_2 2006.

Station number	Station location	Measurement LML (corr)	Calculation Luvotool	RIVM data	RIVM data + façade distance	Luvdata + measurement distance	RIVM data - measurement	RIVM façade - Luvotool	Luvotool measurement distance - measurement
							5 - 3	4-6	7 - 3
1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
136	Heerlen-Looierstraat	42	47	42	43	43	-0.7	3.7	0.6
237	Eindhoven-Noord Brabantlaan	37	57	42	47	51	4.2	10.0	13.7
433	Vlaardingen-Marathonweg	36	34	42	39	36	6.4	-4.7	0.1
448	Rotterdam-Bentinckplein	58	44	48	43	46	-10.0	1.7	-12.3
537	Haarlem-Amsterdamsevaart	41	35	43	48	33	2.0	-13.5	-8.3
639	Utrecht-Erzeijstraat	42	41	45	41	44	3.0	-0.3	2.5
937	Groningen-Europaweg	38	22	33	44	23	-5.3	-21.4	-15.5
	average	**42**	**40**	**42**	**43**	**39**	**0.0**	**-3.5**	**-2.8**
	standard deviation						**5.8**	**10.7**	**10.0**

than overestimates. The use of generic input is a strongly dominant cause of the average deviation and the magnitude of the relative standard deviation.

Luvotool-LDM: non-urban

The LDM is used to calculate not only motorways, but also all other non-urban roads. However, for the non-urban roads, the policy focus is primarily on the motorways. In addition, the number of air quality hotspots on non-urban roads other than motorways is relatively small. The analysis of the results for the non-urban road network is therefore still limited to the motorways. As previously stated, however, the intensities for the most important thoroughfares (not motorways) have been corrected based on data from regional traffic models.

For motorways, the input comprises traffic intensities and the fractions of heavy and medium-heavy vehicles. The data source for the input of the motorway network is unique in the sense that all models use the same data. The data originate from the Directorate for Public Works and Water Management and in past years have been derived from traffic counts, supplemented with interpolations using calculations. The prognoses are from the same source, calculated with the National Model System (*Landelijk ModelSysteem* - LMS, see Section 3.2.3, *the* EMPARA *roads database*). It is therefore pointless to validate the input data based on comparisons with other models.

Comparison with measurements

However, it is certainly sensible to compare the results from Luvotool with measurements. For this comparison, a measurement point of the LML on a motorway (Breukelen) was used (the only such point), along with four motorway measurement points of the measurement networks from Amsterdam and Rotterdam. The result of the comparison with these five stations is shown in Table 3.20.

Table 3.20 Analysis of the differences between measurements and calculations on motorways, NO_2 2006.

Station number	Station location	Measurement	Measurement corrected	Calculation Luvotool	Luvotool corrected	Difference (µg/m³/m³)	Difference (%)
641	Breukelen-Snelweg	45.9	47.0	43.3	49.8	2.8	6%
DCMR	Overschie-Oost-Sidelinge	51.0	51.9	46.3	40.3	-11.6	-22%
DCMR	A16 Ridderkerk-Hogeweg	49.4	50.3	55.0	55.0	4.7	9%
A'dam-018	A10-Zuid	48.0	48.8	47.4	47.4	-1.5	-3%
A'dam-007	Einsteinweg (A10-West)	60.0	61.4	39.8	56.1	-5.2	-9%
	average					**-2.2**	**-4%**
	standard deviation					**6.5**	**13%**

The measurements have been corrected for the difference between measurement elevation and calculation elevation (approximately 3m and 1.5m, respectively). The LDM calculates concentrations at the midpoints of the 25x25 m grid cells. The measurement points are not always located at the midpoints of the grid cells. A correction was made for this difference in calculation distance. In addition, the calculation distance was corrected for several measurement stations because the location of the measurement point or of the motorway in the database did not correspond exactly with reality. The comparison between Luvotool and measurements was conducted using the corrected data.

Conclusion about calculations for motorways
On average, Luvotool underestimated by several micrograms with respect to measurements. However, the relative standard deviation in the differences is large (standard deviation 7 µg/m³); consequently, considering the available number of comparisons, a significant overestimation or underestimation has not been shown. From the theoretical comparison of Luvotool with other models (see above), however, it can be concluded that Luvotool overestimates the contribution from traffic.

3.4 External safety

Data used

General
Data relating to external safety are obtained from the RIVM Centre for External Safety (CEV). The CEV is a certified department of the RIVM (National Institute for Public Health and the Environment). In a number of cases, the CEV receives and processes data from parties such as the National Aerospace Laboratory (NLR). The CEV is responsible for quality control. Table 3.21 provides a survey of the data delivery by CEV.

Calculation of risk distances

Various models are used to calculate risk distances. The distances used for LPG stations are those in the *Kwantitatieve Risicoanalyse generiek voor LPG-tankstations* (TNO-MEP - R 2001/435), 2001. For airports, the following are used: model calculations (TRIPAC V.2.X), IMU-3 and RANI2004 frequencies.

Table 3.21 Data delivery by RIVM/CEV.

Indicator for PBL product		
Risk source	**LPG stations – location-related risk contours 2007**	**Schiphol – location-related risk contours 2006**
Source of data	Inventory by KMPG/OAG (LPG stations and objects of limited vulnerability. KPMG Sustainability, OpdenKamp Advies Groep. 2003, Amended REVI (External Safety Regulations for Installations). STC 2007, 66, p.13.) relating to the locations of high-risk installations and turnover of filling stations. PR 10^{-6} distances according to legislation (REVI, External Safety Regulations for Installations) 2008.	All flight and route information was provided by NLR, based on Schiphol traffic data and data from the IVW (Transport & Water Management Inspectorate).
Data type	Shape (polygon)	Shape (polygon)
Scale of application	Not generalized	Not generalized
Description of data	Location-related risk contour 10-6/yr for LPG stations	Location-related risk contours for Schiphol, year: 2006
Data result from measurements, model calculations (which model), expert judgement	Inventory + implementation of regulations. The distances used for LPG stations with turnover exceeding 1500m³ are those in Fig. 4.1 in the K*wantitatieve Risicoanalyse generiek voor LPG-tankstations* (TNO-MEP - R 2001/435), 2001.	Calculated with NLR software TRIPAC v.2.x, using IMU-3 model, RANI2004 frequencies, contours generated in Matlab
Year	2007	2006
Update frequency	Once-only	On request
Processes carried out	Data (coordinates) entered into GIS system, turnover linked to distances, contours generated around relevant installations.	None
Reliability		Model constraints
Completeness	Coordinates: approx. 95%; turnover approx. 50%	CompleteComplete
Geometric accuracy	5-30 metres (accuracy of GPS readings)	Calculations on 100x100 metre GRID
Quality control performed	Standard review cf. CEV quality handbook (ISO 9001:2000)	Standard review cf. CEV quality handbook (ISO 9001:2000)
Risk source	**Regional airports – location-related risk contours 2005**	**Safety-risk companies reference date: 31 December 2006**
Source of data	All flight and route information was provided by NLR, based on traffic data from regional airports and data from the IVW (Transport & Water Management Inspectorate).	The RIVM database of installations, (reference date 31/12/2006) and location-related risks and group risks from the latest safety reports.
Data type	Shape (polygon)	Not applicable (n/a)
Scale of application	Not generalized	Not generalized
Description of data	Location-related risk contours for Maastricht Aachen Airport (EHBK), Groningen Airport (EHGG), Lelystad Airport (EHLE) and Rotterdam Airport (EHRD), year of use: 2005	Location-related risk contours for companies required to submit safety reports (safety-risk companies). Safety-risk companies are companies with quantities of hazardous substances above the high limit value in the BRZO (Hazards of Major Accidents Decree).
Data result from measurements, model calculations (which model), expert judgement	Calculated with NLR software TRIPAC v.2.x, using Regional model, Cargo_variant_1 frequencies, contours generated in Matlab	Result of processes carried out
Year	2005	2006
Update frequency	On request	Annual
Processes carried out	None	Digitize fN curves (group risk) and 10-6 contours. fN curves are aggregated. Population within 10^{-6} contours was calculated from population database.
Reliability	Model constraints	
Completeness	Complete	Only for companies for which a safety report was available.
Geometric accuracy	Calculations onto 25x25 metre GRID	5-30 metres.
Quality control performed	Standard review cf. CEV quality handbook (ISO 9001:2000)	Standard review cf. CEV quality handbook (ISO 9001:2000)

3.5 Health impact

3.5.1 Context

Environmental health impact modelling and assessment are crucial analyses tools and are of key interest for the PBL. These models and assessments are explicitly formalized in projects, programmes, policy analysis, scientific networking, and national and international advisory tasks. Within the PBL, traditional environmental health issues are being expanded to include more sustainable development and quality of life and place concepts, health impact and disease burden assessment and psychosocial risk concepts, urban community liveability matters and the integration of ecological, economical and social values.

Risk assessment is an important step in the risk management process. Risk assessment measures or models the probability, magnitude and severity of a potential or actual damage or loss. Environmental health policy is increasingly focusing on – and is driven by – the qualitative and quantitative risk assessment of environmental stressors. Traditionally, environmental risk analyses are based on the exposure and health impact assessment of physical stressors, using metrics like population exposure (total exposure or exceedance of standards) and number of people with certain level of health effects (mortality, morbidity). Recently, metrics like the burden of disease, the monetarization of health effects, and the health benefits have been developed and are being increasingly used in risk evaluations and policy analysis.

The main environmental stressors currently investigated include ambient air pollutants (ambient particles and ozone) and noise. Environmental health impact is qualitatively and quantitatively assessed using exposure and health indicators, burden of disease estimates, and assessments of loss of life years and life expectancy.

The PBL Environmental Health Planner (EHP) will be used for ex-ante and ex-post policy analysis and abatement option evaluations and will connect health and wider impacts to causative environmental factors.

The EHP is also being developed in response to specific requests from the World Health Organization (WHO), the International Institute for Applied Systems Analysis (IIASA) and the PBL-LED team. These parties are asking for guidance on regional morbidity assessments (intra- and inter-countries) and are interested in a RAINS NL health impact application tool that, besides mortality assessments, also allows morbidity assessments.

The EHP is designed as an integrated impact assessment tool with three exposure and health elements:
- Environmental health indicators
- Source emission – exposure modelling
- Health impact assessment tool

3.5.2 Environmental health indicators

Environmental health indicators aim to reflect and communicate the status of environmental quality and exposure and health issues (WHO, 2008). They are tools for monitoring and evaluating current and future environmental quality, exposure and health targets and the effectiveness of policies. In addition, these indicators can be used to perform comparative assessments and to evaluate targets in environment and health action programmes. Environmental health indicators are therefore at the core of the information base because they enable assessments to be made of the environment and health situation and the related progress.

Indicators are used as communication means to various users and target groups such as scientists, stakeholders (including citizens) and policy makers. Indicators could therefore be designed for different target groups and purposes. Indicators can be grouped in different types according to their use in policy analysis and evaluation:
- *Performance indicators* refer to a predefined policy target
- *Efficiency indicators* compare the impact with the costs of policy actions and abatement measures
- *Scenario indicators* compare ex-ante or ex-post policy action with 'business as usual'
- *Trend indicators* compare the environmental health situation over a certain time period
- *State indicators* describe what is happening to the environment and human beings and do not explicitly compare anything

PBL performance, state, trend and scenario indicators
The health impact assessment traditionally addresses the extent of the effects in exposure, morbidity or mortality numbers (e.g. expressed as the number of people with allergy symptoms). These health effect indicators can be used to assess scenarios, trends or to assess the effect at a certain point in time (descriptive nature). Regarding the single health effect indicators, a diversity of health effects from a single stressor via multiple pathway exposure can be expressed as an aggregated metric: disability adjusted life years (DALY).

Most recently, risk perception and acceptance have been introduced as additional risk metrics highlighting people's subjective judgment about the characteristics and severity of a risk, irrespective of its physical nature. In addition, cost-effectiveness and cost-benefit ratios are increasingly used in policy evaluation and decision-making. Risk assessment has therefore become a multi-dimensional approach.

The outcome indicators used by the PBL that are related to performance, state, trend and scenarios in policy analysis and evaluation have been developed as described in Table 3.22. The table also presents the data-modelling instruments related to the indicators. Suggestions for developing additional indicators to address these additional risk metrics are discussed in Section 3.5.6.

Table 3.22 PBL Environmental health indicators used in policy evaluations

3.5.3 Emission-exposure modelling

For the route from noise emission to exposure, Noisetool (discussed in Section 3.2) is used to calculate the dispersion and then the load on the receptor. Noisetool calculates the noise level at the façade. This indicator is also used to determine the exposure-response relationships. The spatial details of the model and the health information are compatible in this regard.

The analog route for air pollution is calculated with RAINS-NL (see below). There are two reasons why the LOK model Luvotool (discussed in Section 3.3) is not used for this purpose. First, Luvotool only became operational very recently, and second, the spatial detail level of Luvotool is much higher than that at which the health information is available. Moreover, the use of RAINS has advantages regarding the international comparability of the studies.

Table 3.22 PBL Environmental health indicators used in policy evaluations.

	Indicator[1]	Unit	Applicability[2]				Instruments[3]		
			performance	state	trend	scenario	PM[4]	Ozone	Noise
Exposure Level	ND_{limit}	# days per yr	□	□	□	□	GCN / Rains-nl; Gridstat	GCN / Rains-nl; Gridstat	Empara
	ND_{target}	# days per yr	□	□	□	□			
	LD_{limit}	Level	□	□	□	□			
	LD_{target}	Level	□	□	□	□			
Exposure	$L_{pop.average}$	Level		□	□	□	GCN / Rains-nl; Expo	GCN / Rains-nl; Expo	Empara
	$L_{age\ class}$	Level		□		□			
	L_{regio}	Level		□		□			
	$N_{expo\ class}$	# inhabitants		□		□			
	$N_{at\ risk}$	# inhabitants		□	□	□			
Mortality	APD	# deaths per yr		□		□	GCN / Rains-nl; Knol / LT / EHP	GCN / Rains-nl; Knol / LT / EHP	Empara; Knol / LT / EHP
	YLL_1	DALY		□		□			
	YLL_L	life years	□	□		□			
	$LLE_{newborn}$	life years		□	□	□			
	LLE_{pop}	life years		□		□			
Morbidity	HA	number per yr		□		□	GCN / Rains-nl; EHP	GCN / Rains-nl; EHP	Empara; EHP
	NI	number per yr		□					
	NP	Number		□		□			
Aggregative	BoD LT	DALY		□	□	□	GCN / Rains-nl; EHP	GCN / Rains-nl; EHP	Empara; EHP
	BoD ST	DALY		□	□	□			

[1] Indicator definitions

ND_{limit}	Number of days above limit value
ND_{target}	Number of days above target value
LD_{limit}	Distance of level (concentration) to limit value
LD_{target}	Distance of level (concentration) to target value
L_{pop}	Population average level
$L_{age\ class}$	Age class average level (children, elderly)
L_{regio}	Population average level per region (provinces, rural/urban/street)
$N_{expo\ class}$	Number of people per class of exposure level
$N_{at\ risk}$	Number of people with exposure level above limit value
APD	Attributable Premature Death, difference in number of deaths with and without exposure
YL	Years of Life: total number of life years during the remaining life for total of population
YLL_1	Years of Life Lost: effect of 1 year exposure
YLL_L	Years of Life Lost: effect of a lifetime exposure for start population
$LE_{newborn}$	Life Expectancy for a newborn: lifespan of an average person starting at zero years under conditions (depicted by hazard rates) representing one year
$LLE_{newborn}$	Loss of Life Expectancy for a newborn: effect of steady-state exposure
$LE_{pop.average}$	Life Expectancy pop.: population average remaining live expectancy
$LLE_{pop.average}$	Loss of Life Expectancy pop.: effect of scenario
HA	Number of people affected Hospital admissions
NI	Number of people affected Incidence
NP	Number of people affected Prevalence
BoD LT	Burden of Disease, Long Term
BoD ST	Burden of Disease, Short Term
[2] Applicability	
performance	indicator referring to policy target
state	indicator describing the situation or effect of one specific year
trend	indicator describing the development during a specific period of years
scenario	indicator describing the difference between two scenarios for a specific year or period
[3] Instruments	
GCN	Generieke Concentraties Nederland 8): Maps of the Netherlands with annual average concentrations of air pollution, obtained by data-assimilation of model and monitoring results (output = grid) (historic data)
Rains-nl	Air pollution dispersion model (output = grid) (prognoses)
Gridstat	tool to obtain grid statistics
Expo	tool to obtain statistics about a grid combined with spatial distribution of population
Knol	RIVM implementation of PAR method
LT	Life Table method, in accordance with CBS (compact version)
EHP	Environmental Health Planner: Life Table method (extended version)
[4] PM	Particulate Matter: PM_{10} / $PM_{2.5}$

8) Generic concentrations for the Netherlands

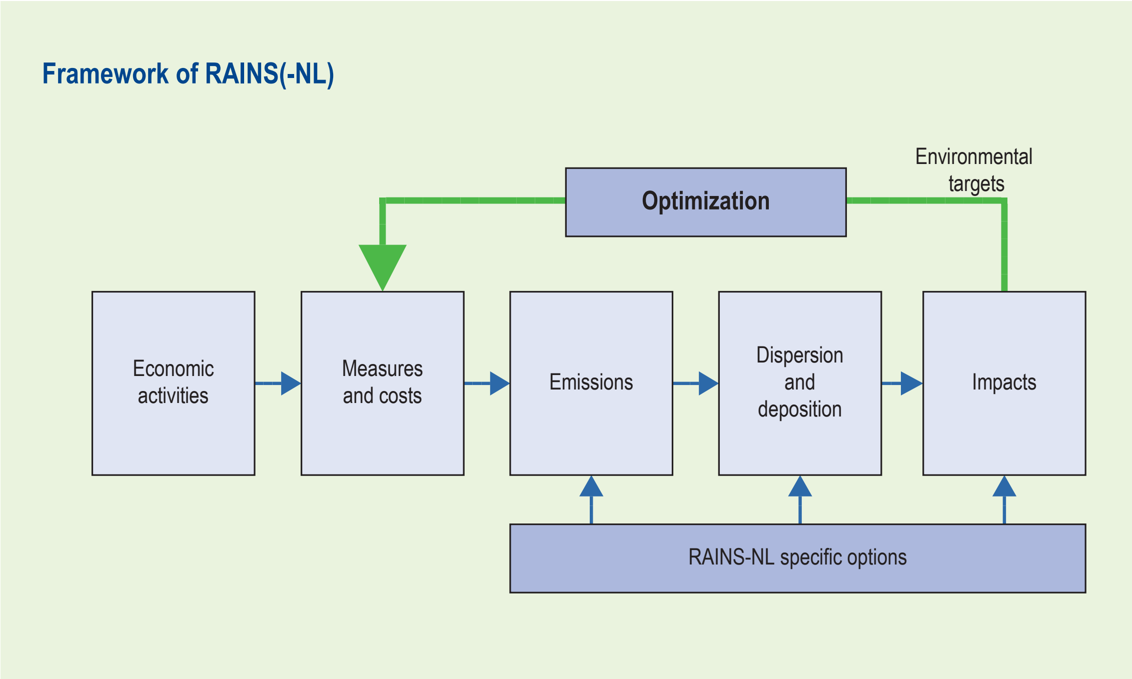

Figure 3.24 Framework of RAINS and the adaption to RAINS-NL.

RAINS - air pollution tool

The **R**egional **A**ir Pollution **IN**formation and **S**imulation (RAINS) model is a tool for the integrated assessment of the impacts of optional emission control strategies. The European version of the RAINS model has been developed by IIASA. The model is used to identify cost-effective emission control strategies at the international level. To be operational at the continental scale, the European version of the RAINS model only includes limited details for each country and thus provides only limited information for the analysis of emission control strategies at the national scale.

RAINS-NL is an adapted version of the RAINS model, developed in cooperation between IIASA and the PBL. This version suits to the specific conditions of the Netherlands and shows more detail relevant for Dutch policy support. The most prominent feature of RAINS-NL is the increased spatial resolution of concentration, deposition and impact. This is achieved by replacing the EMEP-based transfer matrices with a spatial resolution of 50 by 50 km by OPS-based transfer matrices with a resolution of 5 by 5 km.

Benefits of embedding RAINS-NL in RAINS:
- RAINS is the accepted model in European Commission negotiations
- Use of data (activity data, emission factors, costs) is approved by member states
- New scenarios and features in RAINS are immediately available for RAINS-NL

The principal aim of RAINS-NL is to provide a consistent framework to assess emissions, costs, state and impacts. The objectives of RAINS-NL include:
- To support Dutch policy makers in European negotiations. RAINS-NL makes it possible to analyze the implications of European air pollution control policies (CLRTAP; EC-NEC) on Dutch emission control strategies and vice versa.
- To deliver input to cost-effectiveness analysis of abatement alternatives

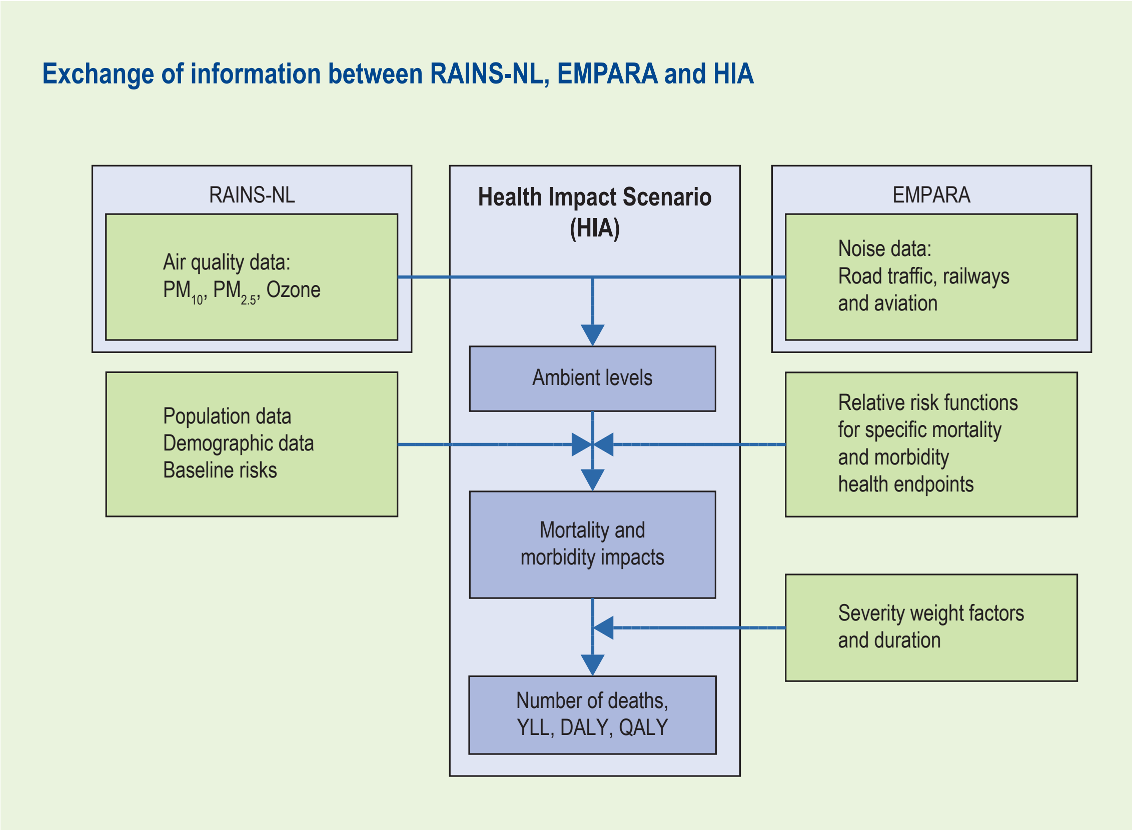

Figure 3.25 Exchange of information between RAINS-NL, EMPARA and HIA, leading to estimates of the health impact of air pollution and noise.

- To provide input to the module that estimates health impacts (mortality, morbidity, and other effects) of air pollution (PM10 / PM2.5 and ozone) expressed as Disability Adjusted Life Years (DALY), including Years of Life Lost (YLL).

3.5.4 Health impact assessment methods and tools

Health impact calculations

Maps produced with RAINS-NL and EMPARA or statistics derived from those maps are combined with population and health data in the health impact assessment (HIA) in order to estimate the health impact of air pollution and noise (Figure 3.25).

Morbidity and mortality figures

The annual mortality risk or the number of deaths as well as the morbidity numbers (based on hospital admissions or doctor visits) related to a certain environment-related disease can be compared with this risk or number in another region or country, or with data from another period in time. It is a function of exposure concentration, population division, exposure-response relationships and the base prevalence of health endpoints.

Population Attributive Risk model (PAR)

The PAR method is used to determine the number of deaths in the population due to exposure to a certain environmental pollutant based on the total number of deaths (D_{totaal}) and the relative risk corresponding with the average concentration found (RRC):

$$D_{PM10} = \frac{RR_C - 1}{RR_C} D_{totaal}$$

Years of Life Lost (YLL)
Years of life lost through premature mortality due to exposure to air pollution is used to estimate how many lost years of life could have been averted in the absence of exposure or with reduced exposure. For each age group, the number of deaths at that age is multiplied by the remaining life expectancy at that age. This is done by obtaining mortality rates from population statistics and exposure estimates.

QALYs
Health impact assessments can also be conducted by calculating the attributable burden of disease. There are several ways to assess the burden of disease attributable to a factor (environmental or otherwise), such as the QALY and the DALY. Quality Adjusted Life Years, QALYs, capture both the quality and quantity elements of health in one indicator. Essentially, time spent in ill health (measured in years) is multiplied by a weighting factor that reflects the relative undesirability of the illness state. This generates a number that represents the equivalent number of years with full health. QALYs are commonly used for cost-utility analysis and to appraise different forms of health care. To do that, QALYs combine life years gained as a result of these health interventions/health care programmes with a judgment about the quality of the gained life years.

DALYs
Disability adjusted life years, DALYs, are comparable to QALYs in that they both combine information on quality and quantity of life. The difference is that QALYs give an indication of the potential number of healthy life years lost due to premature mortality or morbidity and are estimated for specific diseases, not for a health state. Morbidity is weighted for the severity of the disorder. DALYs incorporate three important factors of health: loss of life expectancy due to premature mortality, combined with the duration of living in a deteriorated health state and standardized to the severity of the deteriorated health state. Some DALY calculations also use age weights and time preferences. Age weights indicate the relative importance of healthy life at different ages, for example, an increase of this importance from birth until age 25 and its decline thereafter. Time preference compares the value of health gains today to the value attached to health gains in the future.

DALYs are calculated using the formula:

$$DALY = AB \cdot D \cdot S$$

where $AB = AR \cdot P \cdot F$

and $AR = \frac{RR'-1}{RR'}$

and $RR' = e^{C \ln RR}$

To calculate the adjusted relative risk RR’, information on concentrations (C) in the unit of the relative risk (RR) is necessary. With these data, the attributive risk can be calculated. In combination with the mortality numbers (P) and the fraction of the population exposed (F), this

Table 3.23 Mortality associated with long term exposure to PM_{10}

Duration	Indicator	RIVM	RAINS	CAFE	IOM-LIFET	WHO AirQ
1 yr	# deaths	18.1 thousand	–	15.5 thousand	15.6 thousand	14 thousand
1 yr	YLL_1	178 thousand	–	184 thousand	175 thousand	190 thousand
∞	YLL	–	10.55 million	–	10.16 million	–
∞	LLE	–	0.983	–	1.023	–

results in the attributable burden AB, which reflects the number of people in a certain health state as a result of exposure to the environmental factor that is being analyzed. In the case of air pollution, F is equal to one since everyone is exposed. Disability adjusted life years can optionally be calculated by multiplying the attributable burden (AB) by the duration of the health state, D, times the severity. Severity varies from 0 (healthy) to 1 (death) and therefore in the case of mortality it equals 1. Regarding mortality, the duration of the health state refers to the duration of time lost due to premature mortality, which is calculated using standard expected years of life lost with model life tables.

HIA for Air Pollution in EMPARA and other assessment models

Quantified long-term health impacts of exposure to PM_{10} for the Netherlands have been calculated in a number of impact assessment studies. Several examples of health impact studies on air pollution are discussed in Chapter 2 of this report.

The RR for long-term exposure to PM10 commonly used for the Netherlands is 1.043 per 10 µg/m³ (Künzli, 2000), which is the weighted average from two American cohort studies (Dockery and Pope) and currently the best value that is applicable to the Dutch situation (Knol et al., 2006). This RR is based on exposures to PM_{10} in adults (>30 years) and is therefore is used for members the Dutch population older than 30; all people below 30 years have an RR of 1.

Assuming the reference PM_{10} level of 31.35 µg/m³ in 2000 according to GCN (Large scale concentration maps Netherlands) and the baseline mortality number of 140,527 deaths according to Statistics Netherlands, in 2000 around 17,000 deaths were due to exposure to PM_{10}. According to Knol and Staatsen (2005), 18,100 (12,400-23,800) people are affected by long-term exposure to PM_{10} and are dying on average 10 years earlier (duration). In total, about 181,000 life years are therefore lost in the whole Dutch population (Knol et al., 2005). YLL for PM_{10} were also calculated with different models. A summary of the differences in mortality outcomes with different models is shown in Table 3.23. It can be concluded that the order of magnitude of health effects for these model calculations is tenable.

Appendix 3 contains an additional discussion of the YLL calculations according to different impact assessment models.

Recent exposure-response relationships for morbidity endpoints and mortality from short-term exposure to PM_{10} and ozone were selected based on well-founded Dutch epidemiological studies or, when Dutch data were not available or unsuitable, on international estimates that best suited the Dutch situation (i.e. the exposure range of the Dutch population). Table 3.24 lists the total number of people affected by short-term exposure to PM_{10} and ozone for different health endpoints. (Knol and Staatsen, 2005)

Table 3.24 Total number of people affected due to PM_{10} and ozone in the Dutch population in 2000 (Knol and Staatsen, 2005).

Environmental Factor	Health Outcome	Total number of peaple affected
PM_{10} (short term)	Mortality (total)	1,700 (1,200-2,200) based on individual PM_{10} model. 2,800 (2,200-3,500) based on 2 component-model with ozone.
	Cardiovascular disease mortality	420 (190-660)
	Respiratory disease mortality	580 (430-750)
	COPD mortality	240 (160-340)
	Hostpital admissions cardiovascular disease	2,800 (1,900-3,900)
	Hostpital admissions respiratory disease	700 (430-990)
	Hostpital admissions COPD	500 (340-670)
	Hostpital admissions asthma	not significant
PM_{10} (long term)	Mortality (total)	18,100 (12,400-23,800)
Ozone (short term)	Mortality (total)	1,800 (1,200-2,400) based on individual ozone model. 2,400 (1,600-3,100) based on 2 component-model with PM_{10}.
	Cardiovascular disease mortality	500 (130-870)
	Respiratory disease mortality	not significant
	COPD mortality	not significant
	Hostpital admissions cardiovascular disease	not significant
	Hostpital admissions respiratory disease	not significant
	Hostpital admissions COPD	not significant
	Hostpital admissions asthma	not significant

DALYs

HIA for Noise in EMPARA and other assessment models

Serious effects of exposure to noise
There is increasing evidence of the serious effects of noise on human health. The WHO has mapped out these effects on the European scale (WHO, in preparation). Moreover, a recent European study has discovered a link between aircraft noise and high blood pressure in people living near airports (HYENA, 2008). High blood pressure is a risk factor for myocardial infarct. For road traffic, a direct relationship has been shown between noise and myocardial infarct (Babisch, 2006; RIVM, 2008). For the Netherlands, it is estimated that long-term exposure to high noise levels originating from road and rail traffic leads to about 20-150 myocardial infarcts per year (RIVM 2008).

An explanation for the occurrence of these health effects is given by the evidence [bewijslijst] pointing an underlying stress mechanism that takes effect while experiencing annoyance symptoms and sleep disturbance and that leads to physiological reactions - possibly with cardiovascular disease as a result (TNO, 2007).

A plausible biological mechanism for the relationship between noise exposure and health effects is explained by the hypothesis of stress-induced responses. Physical stress caused by annoyance and sleep disturbance can result in hypertension and myocardial infarct. Several studies in the areas of noise and health and evidence on the construction of the auditive system support this hypothesis. (RIVM, 2008 and Passchier-Vermeer, 1993)

Figure 3.26 Burden of disease expressed in DALYs per million people caused by long-term exposure to PM_{10} 1980 – 2010, Netherlands, with 90% prediction intervals. (Knol and Staatsen, 2005)

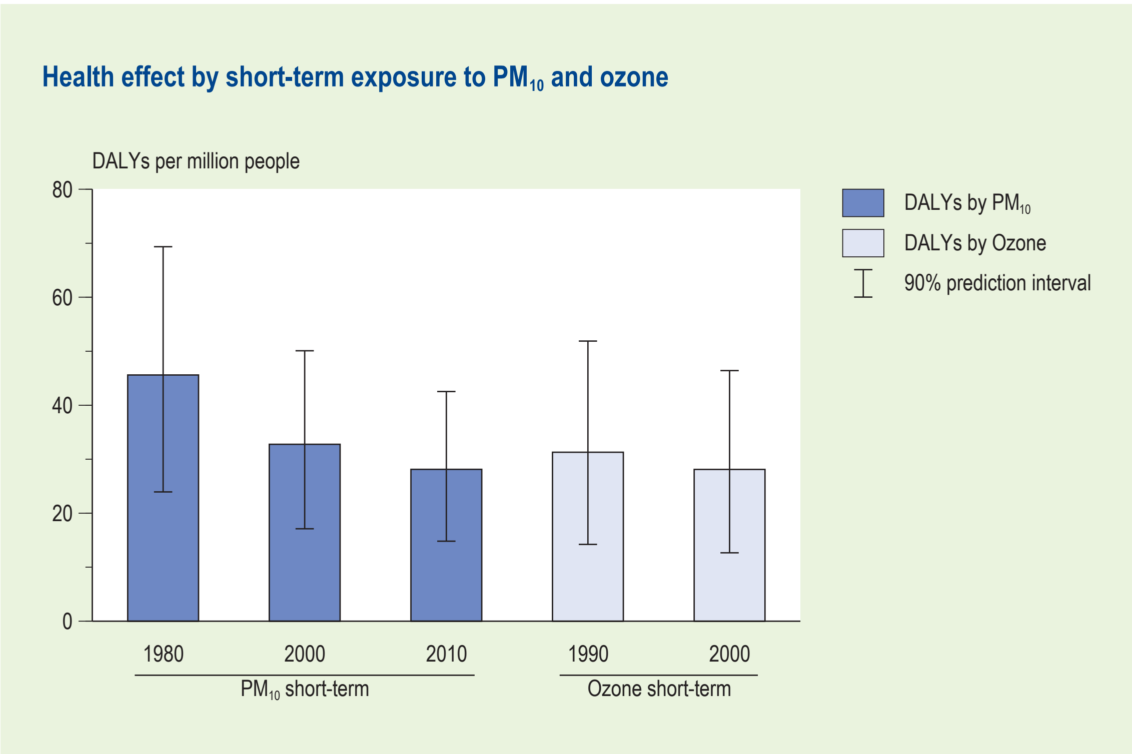

Figure 3.27 Burden of disease expressed in DALYs per million people caused by short-term exposure to PM_{10} and ozone, 1980 – 2110, Netherlands, with 90% prediction intervals (Knol and Staatsen, 2005).

Table 3.25 Health effects from traffic noise in Dutch population (RIVM 2008).

Health effect	Number of persons	95%BI	Indicator	Cut-off point
Acute myocardial infarct per year	84	21 – 150	LAeq, 16h	60dB
Severe annoyance	640,000	480,000 – 830,000	Lden	42dB
Severe annoyance in bedroom	630,000	69,000 – 2,500,000	Lnight	45dB
Severe sleep disturbance[1]	290,000	180,000 – 450,000	Lnight	40dB
Severe sleep disturbance[2]	250,000	12,000 – 2,200,000	Lnight	45dB
At least 13 weekly sleeping problems	86,000	3.4 – 310,000	Lnight	45dB

[1] Exposure-response relationship for road traffic noise derived from Miedema et al., 2002
[2] Exposure-response relationship for road and rail traffic noise derived from Passchier-Vermeer et al., 2007

Population noise exposure is calculated with the EMPARA model. Knol and Staatsen (RIVM, 2005) use source-specific exposure-response functions to calculate the number of annoyed or sleep disturbed people, including noise from road traffic, rail traffic and air traffic (only around the major Dutch airport, Schiphol).

The exposure-response functions for severe annoyance and severe sleep disturbance from transport noise are derived from Miedema et al. (2001 and 2002) and apply to noise levels of L_{den} 42dB(A) (annoyance) and L_{night} 40 dB(A)(sleep disturbance) or above in the adult population. In addition, linear exposure-response relationships were recently derived for night-time exposure to transport noise and the prevalence of severe annoyance in the bedroom, severe sleep disturbance and at least 13 weekly sleeping problems (Passchier-Vermeer et al., 2007).

The HYENA study (2007) assessed the relationships between long term noise exposure from aircraft or road traffic near airports and the risk of hypertension. For night-time aircraft noise, a 10dB(A) increase in exposure was associated with an odds ratio of 1.14 (95%BI: 1.01 – 1.29).

Knol and Staatsen (RIVM, 2005) cumulated mortality numbers of different noise sources. The exposure-response function for cardiovascular mortality was based on the relation between aircraft noise and hypertension and the relation between hypertension and mortality. It was estimated that approximately 600 persons had died from noise-induced cardiovascular mortality. In addition, De Hollander (2004) assessed the prevalence of ischemic heart diseases and premature deaths. The results were similar to Knol and Staatsen (2005). A recent meta-analysis (RIVM, 2008) derived exposure-response functions for hypertension and myocardial infarct which are about a factor 3 lower than the exposure response functions used in Knol and Staatsen (2005) and de Hollander (2004). Case-control and follow-up studies, included in the meta-analysis, led to this lower estimate. A statistically significant exposure-response was only demonstrated for road traffic noise exposure and myocardial infarct: for each 5dB(A) increase in exposure, a 6% increase in risk was found (95% CI: 1.01 – 1.11), resulting in approximately 84 noise-induced myocardial infarcts. Table 3.25 summarizes the latest findings on health effects from road traffic noise in the Dutch population.

DALYs

Knol and Staatsen (2005) calculated DALYs for a number of noise-induced health effects for the years 1980, 2000 and 2020, which are presented in Figure 3.28.

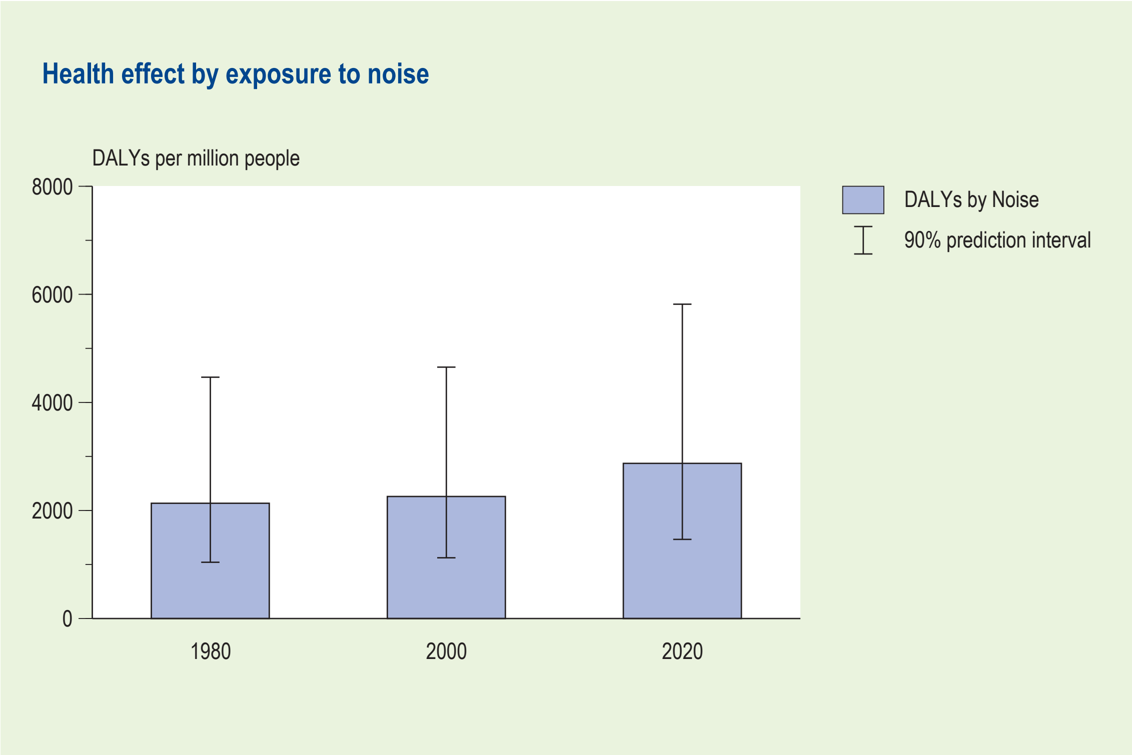

Figure 3.28 Burden of disease in DALYs per million people caused by exposure to noise, 1980 – 2020, with 90% prediction intervals (Source: Knol and Staatsen, 2005).

In 2000, 400 DALYs per million people could be attributed to noise-induced cardiovascular diseases. The choice of exposure-response functions for noise-related annoyance and noise-related sleep disturbance largely affect the DALY estimates for these health end points, which is shown in Table 3.26.

3.5.5 Uncertainties

Health impact assessments entail a number of uncertainties. These concern input data, assessment methodologies (not only due to accepted and objective calculation formulas but also to subjective expert judgments, inimitable and otherwise), and interpretations of assessment outcomes. In addition, for much data, predictions have to be made, if possible, about what will change in the future. Due to these limitations, the results of assessments of the health impact of environmental stressors should be seen as indicative, and not necessarily the truth. The assessment outcome should always be viewed in the context of the degree of uncertainty, assumptions and confidence.

Monte Carlo analysis indicates the potential range of outcomes. This method uses the uncertainty ranges based on literature for the exposure, RR, severity factor and duration. With this method, a 90% prediction interval is estimated around the output. All variables are assigned a series of random values within their range, thereby estimating the range of the output distribution. In this type of simulation, probability distributions are defined for each uncertain variable.

Air pollution (PM_{10}, $PM_{2.5}$ and ozone)
In CAFE, the WHO made a systematic review of the health impact of ozone and particulate matter (WHO, 2003 and 2004b). The most important conclusions were the following:

Table 3.26 DALYs for severe noise annoyance and severe sleep disturbance, based on exposure-response curves (Miedema, 2001) and on a survey (Franssen et al., 2004) (Source: Knol and Staatsen, 2005)

		DALY's per million people	
		Miedema exposure – response curves (Miedema, 20021), year: 2000	Environment-related annoyance and quality of life survey (Franssen, 2004), year: 2003
Severe annoyance	Road	1,122 (441-2,753)	7,604 (3,119-18,387)
	Air	16 (6-38)	314 (129-761)
	Rail	65 (24-158)	524 (215-1,268)
Severe sleep disturbance	Road	526 (189-1291)	3045 (1,298-7,029)
	Air	-	761 (324-1,757)
	Rail	32 (10-80)	253 (108-586)

- There is no threshold for particulates and ozone below which no adverse effects are expected.
- Current limit values do not preclude adverse health effects, and further reductions in air pollution will have significant health benefits, even in regions where levels are well below current limit values.
- Quantification of the contributions from different sources and of different particle components to health effects is currently impossible. However, particulate matter from combustion sources seems to be especially related to adverse health effects.
- Many studies found that particulate matter ($PM_{2.5}$) has serious effects on health, but that coarser particulates (between PM_{10} and $PM_{2.5}$) may also have adverse health effects.
- The long-term effects of $PM_{2.5}$ clearly outweigh those of the short-term effects.

Effects from short-term exposures

There are currently more than a hundred time-series studies on the acute effects of exposure to particulate matter being conducted on most of the continents. These have established an association between air pollution and mortality and/or morbidity in quite different populations (as measured by their gross domestic product (GDP) or purchasing power parity (PPP)) and for very diverse exposures to quite different mixtures of PM.

In the Netherlands, health impact assessments have been made for acute effects associated with short-term exposure to ozone and particulate matter based on Dutch data. The most recent estimate shows that a few thousand people died several days to several months prematurely in the Netherlands in 2003 (Fischer et al., 2005). Of these mortalities, about 2,300-3,500 are attributed to particulate matter and 1,100-2,200 to ozone.

Effects from long-term exposures

The scientific basis for chronic effects of particulates is small, and there are currently only four studies available for health impact analysis with results of the full cohort. The composition of these four American cohorts is mixed: some cohort studies involve the general population, such as the six cities study by Dockery et al., (1993) and the American Cancer Society (ACS) cohort by Pope et al. (2002), while other studies are limited to specific groups, such as the Seventh Day Adventists (Abbey et al., 1999) or retired US army personnel with high blood pressure (Lipfert et al., 2001). The Pope study used in CAFE is the largest cohort study published in the scientific literature. Despite a number of uncertainties surrounding the ACS cohort and some objections against the direct use of values from Pope et al. (2002) for quantitative estimates in Europe (see textbox) assessments of the health impacts of particulates are often based on this study.

Use of the Pope study

There are a number of uncertainties surrounding the ACS cohort and some objections against the direct use of values from Pope et al. (2002) for quantitative estimates in Europe. These concerns are discussed below.

- The smoking status in the cohort was only assessed once – at the beginning of the study – while smoking habits in the USA have changed considerably since that time; such changes in smoking habits most likely have a social economic status (SES)-related component as well and might confound the observed frequencies of health effects in the population of the ACS;
- No individual-based exposure is assessed in the ACS cohort; instead, city average exposures are used as a proxy for this individual exposure assessment;
- A higher SES seems to mitigate the health impact of particulate matter in the ACS cohort, which may point to an as yet unaddressed confounding or smoking-related influence (compare with first item);
- The application of American results indiscriminately to Europe ignores the existing differences in air pollution mix and sources as well as the population health status; for example, the use of a different proxy (sulphate) from the same cohort as a measure for the air pollution mixture leads to a negligible health risk in the Netherlands;
- The quantitative effect of air pollution (RR) concerning heart problems or lung cancer changes during the follow-up periods in the ACS cohort.

However, two other studies show either no effect or only an effect on men, introducing questions on the strength and direction of the relationship between particulate matter and long-term health effect.

If specific results from Pope et al. (2002) are applied to the Netherlands, more than 10% of the Dutch population could be assumed to have died 10 years prematurely due to long-term exposure to ambient particulate matter in 2000. This corresponds to a loss in life expectancy of about 1 year when averaged over the whole Dutch population or to approximately 18,000 people dying a decade sooner (Knol and Staatsen, 2005).

NO_2 as a surrogate index

Although epidemiological studies have found associations between health and NO_2 values around the current limit value (WHO, 2004a), there generally is serious doubt about whether exposure to NO_2 itself is the cause. It is believed that in these situations ambient NO_2 should be seen as an indicator of traffic-related air pollution. This mixture is associated with health effects, as also suggested by the results of Krämer et al. (1999).

RAINS and CAFE

In CAFE, only mortality associated with chronic exposure to particulate matter has been calculated. The RAINS calculations of loss of life are based on an American study carried out by Pope et al. (2002), while the Dutch study is based on a combination of an American study carried out by Dockery et al. (1993) and the one by Pope et al. (2002) (Knol and Staatsen, 2005). Since both CAFE and the Dutch health impact analyses are based to a certain extent on the same studies, it is no surprise that both assessments deliver very similar health impacts in quantitative terms.

Because an overestimation of ambient anthropogenic particulate matter concentrations in RAINS Matthijsen and Brink, 2007), the estimated improvement of the health impact by abatement measures in the Netherlands will be lower than that calculated with RAINS. RAINS attributes all of the health effects to anthropogenic $PM_{2.5}$, whereas Knol and Staatsen (2005) attribute the health effects to the total PM_{10} concentration. The Dutch calculations indicate that estimated total PM_{10} levels – and the health impacts that are 35%, respectively, compared to 2000. These values are significantly lower than the relative 27% and 52% that RAINS calculates for the same periods. However, as it is unknown which fraction of the particles causes the health effects, both approaches may be valid, or partially valid, or some totally different scenario may turn out to be more cost-effective if one of the alternative hypotheses for the causal fraction(s) eventually receives more scientific support.

Uncertainties in the health impact assessment data base
Although the respiratory effects that result from exposure to ozone have been clearly demonstrated, the biological mechanisms, the exact sources and the causal fraction(s) responsible for the effects of exposure to particulate matter are as yet largely unknown. The relationship between short-term changes in air pollution and health effects in a population has been found in several studies, whereas to date there have only been five studies on health effects from long-term exposure to particles – with partly contradicting results. The data base on health effects from long-term exposure to ozone is currently too limited to base firm conclusions on.

The current level of knowledge about the health effects of long-term exposure to particulate matter is low and the uncertainties in the above assessments are therefore greater. The small number of long-term effect studies about particulate matter and their sometimes conflicting results are the most important causes of this low level of knowledge. The level of knowledge concerning the effects of short-term exposure is higher, and the uncertainties are smaller.

The uncertainties in the risk assessment primarily concern the following issues:
- the question of whether or not the observed statistical correlation from the epidemiological research indeed has a cause-effect relationship, if the correct particulate matter indicator was used and if there has been sufficient correction for other distorting variables;
- the question of whether research data from other countries can be applied to the exposure situation in the Netherlands due to differences in the population, the composition of the particulate matter and the other aspects of air quality;
- the assessment of the magnitude and duration of the various effects;
- the statistical uncertainties in the assessment of the risk factors;
- the question of whether the relationships found are indeed linear;
- the decision of whether or not to use a threshold value and revert to a hypothetical concentration without any particulate matter in the outside air.

Noise

Uncertainties about the exact noise level for serious effects
There is a high level of uncertainty about the exact noise level at which serious health effects occur, the so-called effective noise level. Regarding serious health effects due to long-term exposure to noise, the WHO concluded in 1999 that these effects occur at noise levels of 65-70 dB(A) or higher. However, there are indications that these effects can also occur at lower noise levels; this is because epidemiological research has been unable to ascertain a noise level at which health effects do not occur. The RIVM concluded that the choice of the effective noise level has a major influence on the assessment of the magnitude of the serious effects and the corresponding disease burden (RIVM 2005 and 2008). For example, the estimated number of myocardial infarcts for the Netherlands referred to above would be increased by a range of approximately 90-700 if a lower effective noise level of 55 dB(A) is assumed (RIVM, 2008). This can be explained by the fact that a larger group of people in the Netherlands would be exposed to this lower noise level.

Table 3.27 Cut-off points for exposure-response relationship of myocardial infarct due to road traffic noise and consequences for input estimates.

Exposure-response relationship assumed from: (dB(A))	Conversion from Lden toLAeq,16h according to situation:			
	Urban		Motorways	
	Number of people	95% confidence interval	Number of people	95% confidence interval
55	240	58-430	180	45-330
60	53	13-94	36	9.0-65
65	6.0	1.5-11	3.5	0.86-6.1

Choice of exposure indicator
Noise exposure can be reported with different noise level indicators. The $L_{Aeq, 16h}$ is used as exposure indicator for deriving the relationship between road traffic noise and myocardial infarct. Using this indicator excludes night-time exposure from the health impact assessment. However, the Health Council of the Netherlands concluded in 2004 that a correlation between night-time exposure and increased risk of hypertension and cardiovascular diseases is plausible. A causal relationship is yet to be proven.

Shape of the exposure-response relationship
The shape of the exposure-response relationship for noise is subject to uncertainty. The small RRs for noise levels below 65dB(A) do not allow for defining a no-effect level for myocardial infarct mortalities. The cut-off point can significantly influence the health impact estimates (RIVM, 2008).

Study design
Cardiovascular diseases and deaths from traffic noise exposure are studied only in small population groups, including mainly middle-aged men. A number of studies involving children demonstrate inconsistent hypertension effects from traffic noise exposure.

Most studies on health impacts from noise exposure are cross-section studies. Recent case-control and follow-up studies result in more valid estimations on the order of health effects.

When comparatively analyzing health effects from different studies, the study design and study parameters require careful and close inspection. For example, "*Hinderinventarisatie*" (survey of annoyance) compared to Miedema (2002).

Co-mortality effect
The biological mechanism underlying the incidence of myocardial infarct can explain the possibility of a co-mortality effect from noise and air pollution exposure. Both environmental pollutants may lead to stress responses, which may then result in cardiovascular diseases (via natural resistance reduction or behavioural responses such as increased smoking).

3.5.6 Current and future EHP developments

RIVM Chronic Disease Model - integral assessment of public health
From the viewpoint of public health, health status can be interpreted as the outcome of a multi-causal process with various determinants. The conceptual scheme in Figure 3.29 places public health in the centre of four groups of determinants: (1) endogenic or person-related

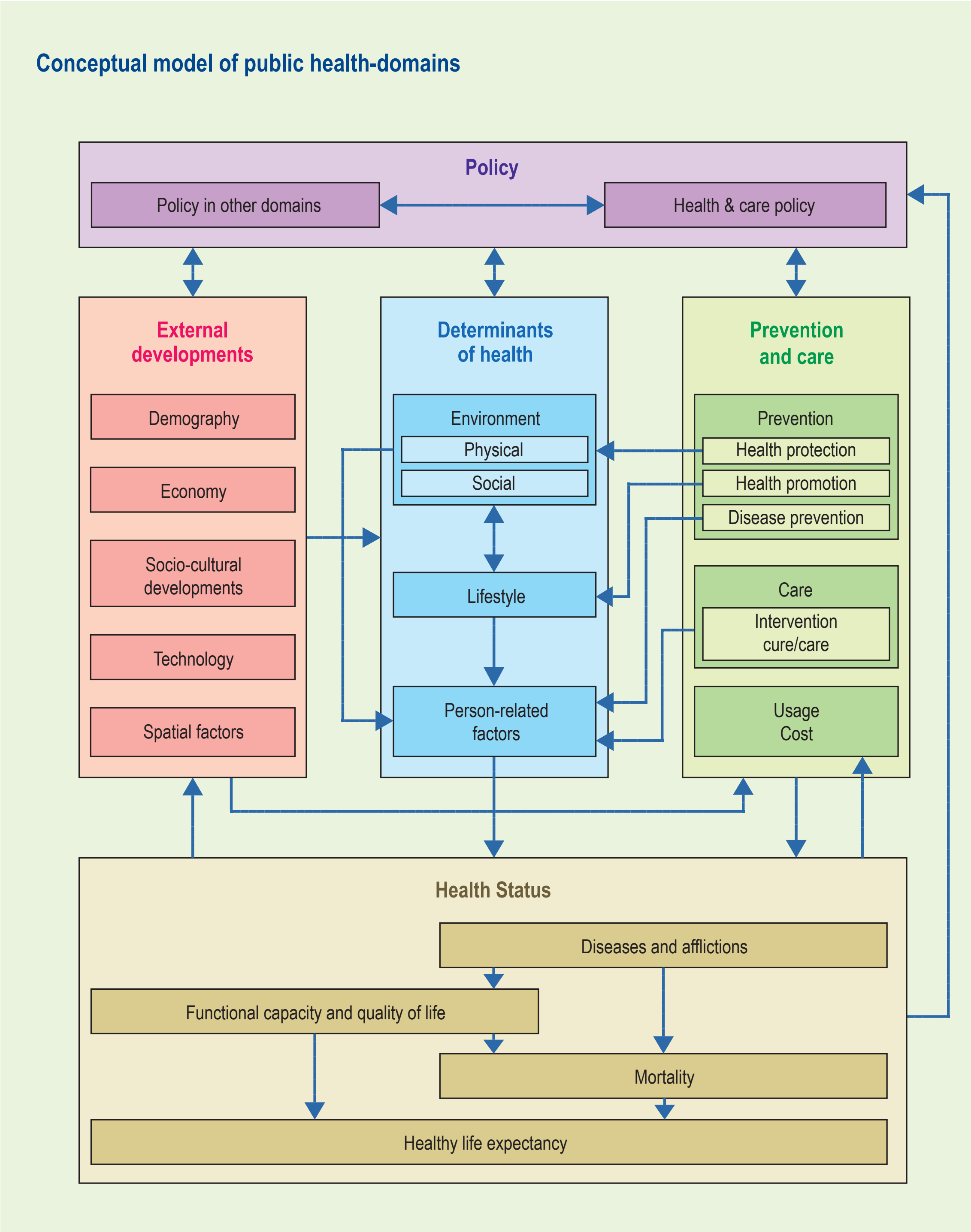

Figure 3.29 Conceptual model of public health - domains, determinants and outcome. (Source: RIVM (2006) Care for health. The 2006 Dutch Public Health Status and Forecasts Report.)

characteristics (genetic, biological), (2) lifestyle, (3) physical and social environment and (4) health care (including preventive action).

Insight into the interaction between environmental determinants and other risk factors can be gained by using a model covering all four groups of determinants. The RIVM Chronic Disease Model (CDM) is a dynamic population model that was developed to estimate the effects of

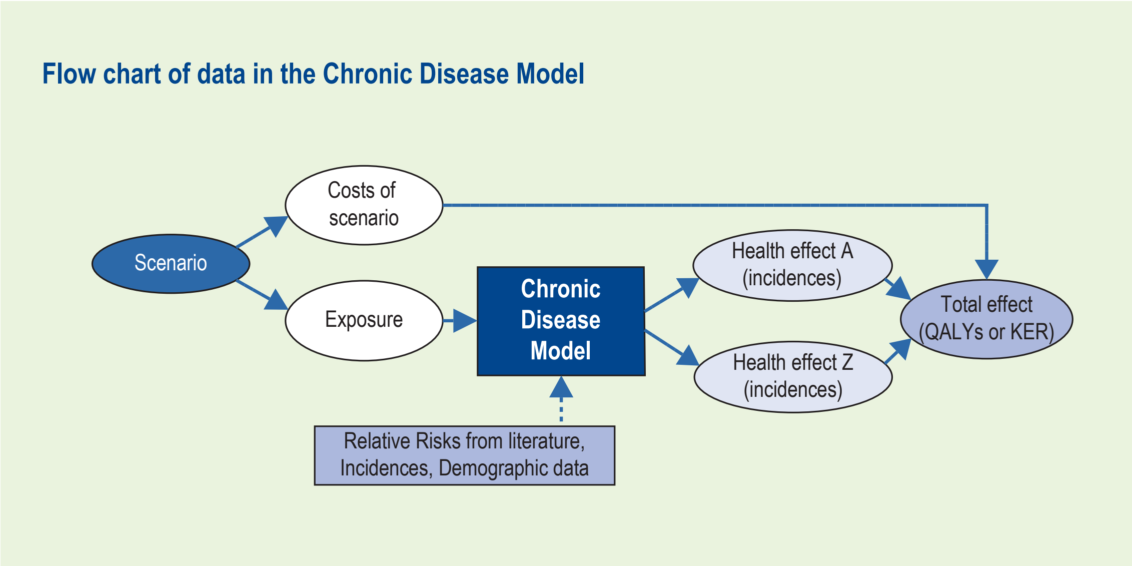

Figure 3.30 Flow chart of input and output data in the RIVM Chronic Disease Model.

changes in the prevalence of risk factors (such as smoking and overweight) on the prevalence of chronic diseases and the resulting mortality in the Netherlands. This model is used for two scenarios, one with changes and one without. The effects are represented by the differences in the occurrence of chronic diseases and mortality and finally the number of (healthy) years lived in each simulation run (see Figure 3.30).

A more realistic result is expected when also environmental risk factors are integrated with life-style risk and person-related factors.

The CDM) may also be used as an alternative for the monocausal, population attributable risk assessment approach by integrating (competitive) environmental causes for diseases into one integrated analysis. A brief description of the model is given below. (Baal et al., 2005)

Description of CDM

CDM has been developed as a tool to describe the morbidity and mortality effects of autonomous changes of and interventions on chronic disease risk factors, while taking integrative aspects into account. The model contains several risk factors including cholesterol, systolic blood pressure, smoking, activity level and Body Mass Index. It models 28 chronic diseases: cardiovascular diseases, distinguishing acute myocardial infarction, other coronary heart disease, stroke, and chronic heart failure, COPD, asthma, diabetes mellitus, dementia, osteoarthritis, dorsopathy, osteoporosis and 15 different forms of cancer. The model is structured in such a way that new diseases and risk factors can be added relatively easily. The mathematical model structure is called a multi-state transition model and is based on the life table method. The model states defined are the risk factor classes and disease states. State transitions are possible due to changes between classes for any risk factor, incidence, remission and progress for any disease, and mortality. The model describes the life course of cohorts in terms of changes between risk factor classes and changes between disease states over the simulation time period. Risk factors and diseases are linked through relative risks on disease incidence. This means that incidence rates for each risk factor class are found by multiplying the relative risks by the baseline incidence.

The main model parameters are:
- the initial population numbers;
- initial class prevalence rates and transition rates for all risk factors;
- initial prevalence, incidence, remission and progress rates for all diseases and;
- relative risk values specified by risk factor and chronic disease.

All model parameters and variables are specified by gender and age. The time step used for modelling is 1 year. The main model outcome variables are incidence, prevalence and mortality numbers specified by disease, and integrative measures such as total and quality adjusted life years. Examples of the integrative aspects of the model are the joint effects of combined risk levels (where different causes of morbidity and mortality are distinguished) the effects of mortality selection and the statistical modelling of dependent competing risks.

First tests with CDM with inclusion of PM_{10} as determinant
RIVM-CDM is used to model the effects of exposure to PM_{10} on a population without births and migration. The composition of the start population is that of the Netherlands in 2005. The situations modelled are those with and without exposure to PM_{10} for a period of 110 years, until the entire start populating has died. By definition, the number of deaths is exactly equal to the start population. The difference between a population with and without exposure is expressed in the number of life years per year and in total. The effect of exposure is modelled in two ways: the effect of the year 2005 only (1 year exposure) and the effect of a continued reduction (remaining lifetime exposure). In both cases the PM_{10} is reduced to 0-level, which is not realistic because of existing natural background and a foreign contribution. Another assumption is the absence of a time-lag.

The following indicators are used. For 1-year exposure, the difference in years of life (YLL_1) and the number of attributable premature deaths (APD); for lifetime exposure, the difference in years of life (YLLL) and the difference in the length of the average life span (LLE), which is a weighted average over all ages in the start population. Table 3.28 shows the effects attributed to PM10 and, for comparison, the effects of smoking.

Table 3.28 Mortality and LLE (years) in the chronic disease model

The calculation with 1-year exposure represents the effect due to exposure to a single year. The results show that the effects of PM_{10} and smoking, individually or simultaneously, behave differently. The results for PM_{10} and smoking are additive for lifetime exposure, while they are not for 1-year exposure.

Towards more regionalized impact assessment and policy evaluation
Regionalized information on emissions and concentrations as well as health effects allows policy measures to be taken on a smaller, sometimes even urban scale, which may result in a larger health impact. This is because it is known where the concentration of pollutants is highest and the health impacts largest.

Current PM_{10} concentrations are known on a very small spatial scale (1 x 1 km grid). It would be interesting to compare this with health effects that are known to be related to exposure to PM_{10}. Although mortality data are widely available at the municipal level, morbidity data are generally not available at this level. Therefore, DALYs usually have to be calculated on the national scale at present. Alternatively, when opting for a smaller scale, not all relevant health effects can

Table 3.28 Mortality and LLE (years) in the chronic disease model.

Intervention		Health indicators (saved lives or years)			
PM_{10}	smoking	YLL_1	APD	YLL_L	LLE
✓	✗	168,000	14,281	$17.7 \cdot 10^6$	1.09
✗	✓	340,000	28,875	$31.6 \cdot 10^6$	1.94*
✓	✓	471,000	40,049	$49.3 \cdot 10^6$	3.03

* average over entire population

YLL_1 years of life lost – 1year exposure
APD number of attributable premature death
YLL_L years of life lost – lifetime exposure
LLE loss of life expectancy

be included and therefore the health impact is likely to be underestimated, especially since the prevalence of certain morbidity effects (like coughing and breathing troubles due to exposure to ozone) is very high.

Morbidity endpoints

Appendix 4 includes a comparison of the available exposure-response relationships for mortality and morbidity endpoints from long-term and short-term $PM_{2.5}$ and PM_{10} exposure. The relevance and choice of the exposure-response relationships for morbidity endpoints for use in Dutch assessments is still being decided. For noise morbidity endpoints, the selected exposure-response relationships as listed in the meta-analysis of RIVM (2008) can be applied.

Sensitivity and importance analysis

The estimation of the health effects of environmental stressors always involves uncertainty. It would be interesting to be able to quantify the effect of uncertainty on the PBL health impact assessment model results and to compare the relative importance of the different input variables and model assumptions. One application of uncertainty analysis in life-table modelling is described in Tainio, et al. (2007)

The PBL approach to additional risk metrics

The desirability and fairness of equal protection of citizens against risks from environmental factors are increasingly being weighed against an effective commitment to spend the limited collective means for reducing risks. As part of policy assessment, other aspects besides the physical magnitude of health effects are also considered. These include the economic valuation (cost efficiency, cost-benefit ratio), the perception of risks (subjective or otherwise) and the acceptance of risks. Consequently this concerns not only '*magnitude',* but also '*money and emotion*'.

During this process, 'Risk perception' is defined as 'not only the entirety of beliefs, judgments and feelings, but also the social and cultural norms and values that people have with respect to risks'. This consequently concerns the 'subjective assessment of the probability of the occurrence of a specific event, the degree with which this causes unrest and the extent with which people are prepared to accept this unrest'.

If risk perception and acceptance are to be included in policy evaluations, also as part of risk comparisons, it is important to acknowledge the characteristics of risks and how they are evaluated (scored) by the population. To obtain insight into this process, the following questions are important:

1. What risks are important for the population?
2. How concerned is the population about these risks?
3. How acceptable are these risks for the population?
4. Which risk aspects are important in the subjective evaluation of risks?

At the PBL, the research into the experience of risks is being conducted by the RIVM (R. van Poll et al.). Two methods are currently important in this process: the ‘psychometric paradigm’, which provides insight into the relevant risk perception aspects and the way in which they can be quantified, and the ‘multi-attributive use theory (MAUT), which provides structure and techniques to compare different types of risks and to evaluate them based on the relevant, underlying risk aspects.

These methods are being used in the project *‘Nuchter omgaan met risico’s II*’ (Dealing clearly with risks II); the corresponding analyses are currently taking place.

4 Dealing with uncertainties

In its description of the methods, Chapter 3 indicates which uncertainties play a role in the various dossiers and how large these uncertainties are. This chapter describes how the PBL deals with uncertainties in the policy evaluation.

Scientists are accustomed to dealing with uncertainties in the results of their research. The results are often linked to confidence intervals. However, policymakers usually cannot deal very well with such uncertainties. They want to make choices based on clear figures. This leads to the tendency to demand increasingly precise conclusions from scientists. As a result, they are asked to stretch the limits of their knowledge too far and there is a risk that they will be swept along in a process of false certainty. It is therefore crucial that scientists and policymakers work together to establish a picture of uncertainty and how to deal with it.

4.1 Air traffic

During the discussions on the research and assessments of noise exposure and external safety risks, the aspect of uncertainty is addressed in two ways. Firstly, there is a consensus among all parties involved with this dossier about the use of a single model for both research and policy evaluation. The results of this model are assumed to be valid, which avoids disagreements. Secondly, open communication with the stakeholders about methodological aspects ensures that as many of them as possible understand and accept the existence of uncertainties.

Consistent use of models

The fact that the computer models generate the same results does not mean that they are also 'accurate' in the sense that they provide a correct representation of reality. It should be noted here that 'accuracy' in this case cannot be seen separately from the uniform and consistent use of certified models – not only by research institutes, but also by all policy agencies. This applies, for example, to the research that was conducted beginning in the 1990s as part of the Schiphol Airport Health Assessment (referred to by the Dutch abbreviation GES). The methods for assessing health effects and risks that resulted from the GES research are also used by the PBL (Breugelmans et al., 2005).

The consistent use of a single model for both research and policy (policy making and policy evaluation) ensures that the possible measurement uncertainty in the model plays a subsidiary role. During both policy enforcement and the use of assessment methods for health effects, a 'balancing effect' occurs, which tends to cancel out possible systemic errors and measurement uncertainty.

Methodological agreements

Before and during the research, discussions were conducted with a large number of parties including the relevant ministries, Schiphol Airport, the KLM, Air Traffic Control the Netherlands, residents' organizations and universities and research institutes. During these discussions, many issues emerged, some of which are listed in chapter 4. These issues are also addressed in a number of articles published in professional journals (such as Dassen, 2006; Dassen, 2007; Dassen 2008).

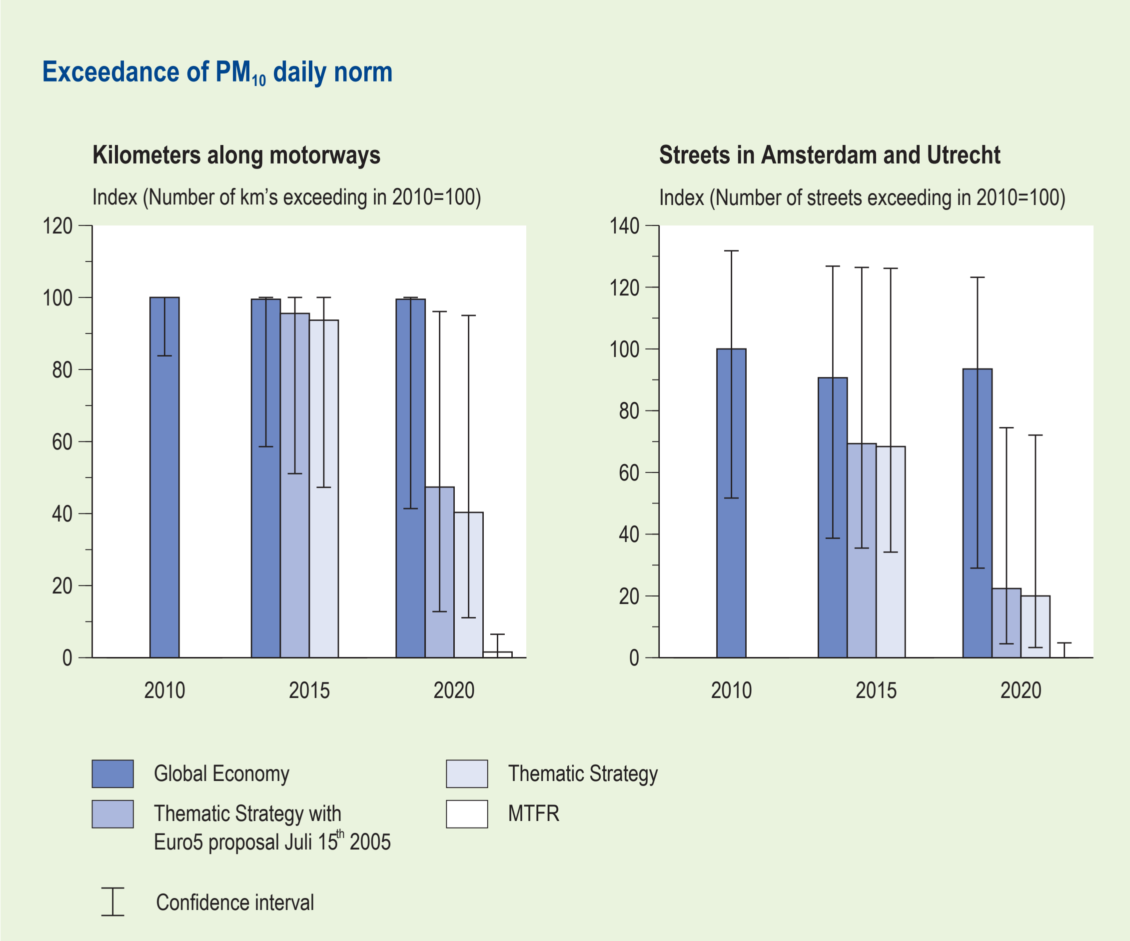

Figure 4.1 Number of exceedances of the PM10 daily limit value along motorways (km) and streets in Utrecht and Amsterdam with respect to current policy 2010 (Global Economy), for thematic strategy, thematicstrategy with Euro 5 proposal 15/07/05 in 2015 and 2020 and Maximum Technical Feasible Reduction (MTFR) for 2020. The range indicates the confidence interval of 33-66%. Source: Folkert (ed.) (2005).

The report *Het milieu rond Schiphol, 1990-2010, Feiten en Cijfers* (PBL(a), 2005) (The Environment around Schiphol, 1990-2010, Facts and Figures) focuses explicitly on a large number of methodological issues and how these issues are dealt with.

4.2 Air quality

The policy evaluations refer as much as possible to the sometimes substantial uncertainties in the calculations on which the conclusions are based. Conclusions about local air quality are made only when this concerns a large area (such as a city) that includes a sufficiently large number of situations to which the conclusion applies. No conclusions are made about specific, local situations. Where possible and relevant, a bandwidth is included in the results, see Figure 4.1.

In the air quality dossier, the current discussion on uncertainties is taking a different direction than that in the air traffic dossier. Developments in this dossier have resulted in the rejection of plans for construction projects on legal grounds, sometimes involving alleged exceedences of air

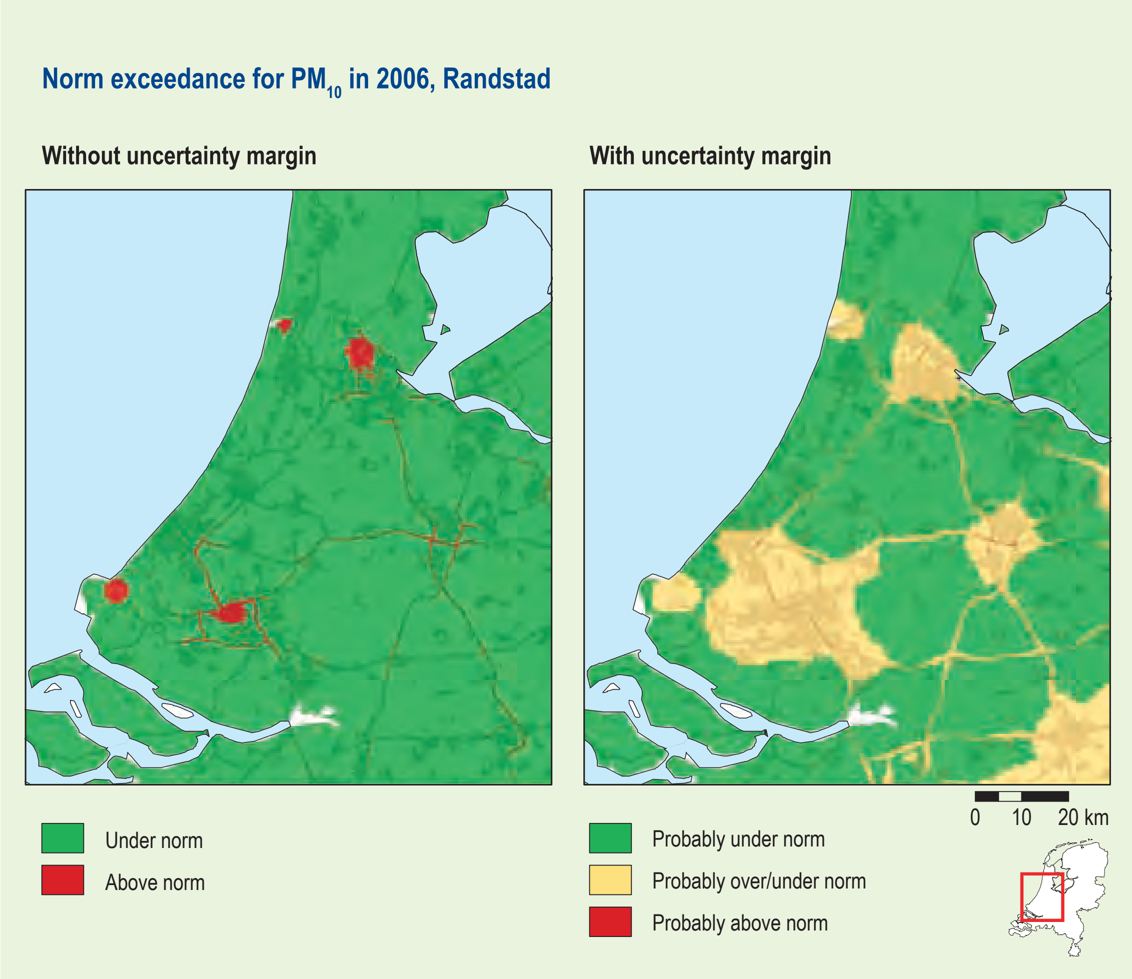

Figure 4.2 Illustration of areas with norm exceedances in the Randstad, without an uncertainty margin (left) and with an uncertainty margin (right). The figure on the left shows whether or not the norm is exceeded based on a median estimate, without taking account of the uncertainties in this estimate.

quality norms in the range of tenths of micrograms (see also Section 2.3. This absolute interpretation of such concentrations is disproportional to the uncertainty of the concentrations, which often amounts to many micrograms.

For the PBL, this was a reason to present a method to policymakers for dealing with uncertainties. Due to the uncertainties, there is a 'grey' area of several micrograms surrounding the norm. Within this range, no unequivocal conclusions can be made about norm exceedences in a scientifically responsible fashion. Outside this grey area, conclusions can be made about exceedences with a well-defined uncertainty. Based on the uncertainties that are established in policy, the grey area can be defined in a scientifically responsible fashion. Within this grey area, the continuation of construction activities or the necessity to take measures to improve air quality become governmental choices that depend on more factors than air quality alone. Figure 4.2 shows an example of the consequences of this approach.

Figure 4.3 is the same as Figure 4.2, but shows an enlargement of an area surrounding the A10 motorway near Amsterdam-West. This figure illustrates that the different approaches can also lead to other policy measures. Based on the left figure, authorities could consider installing

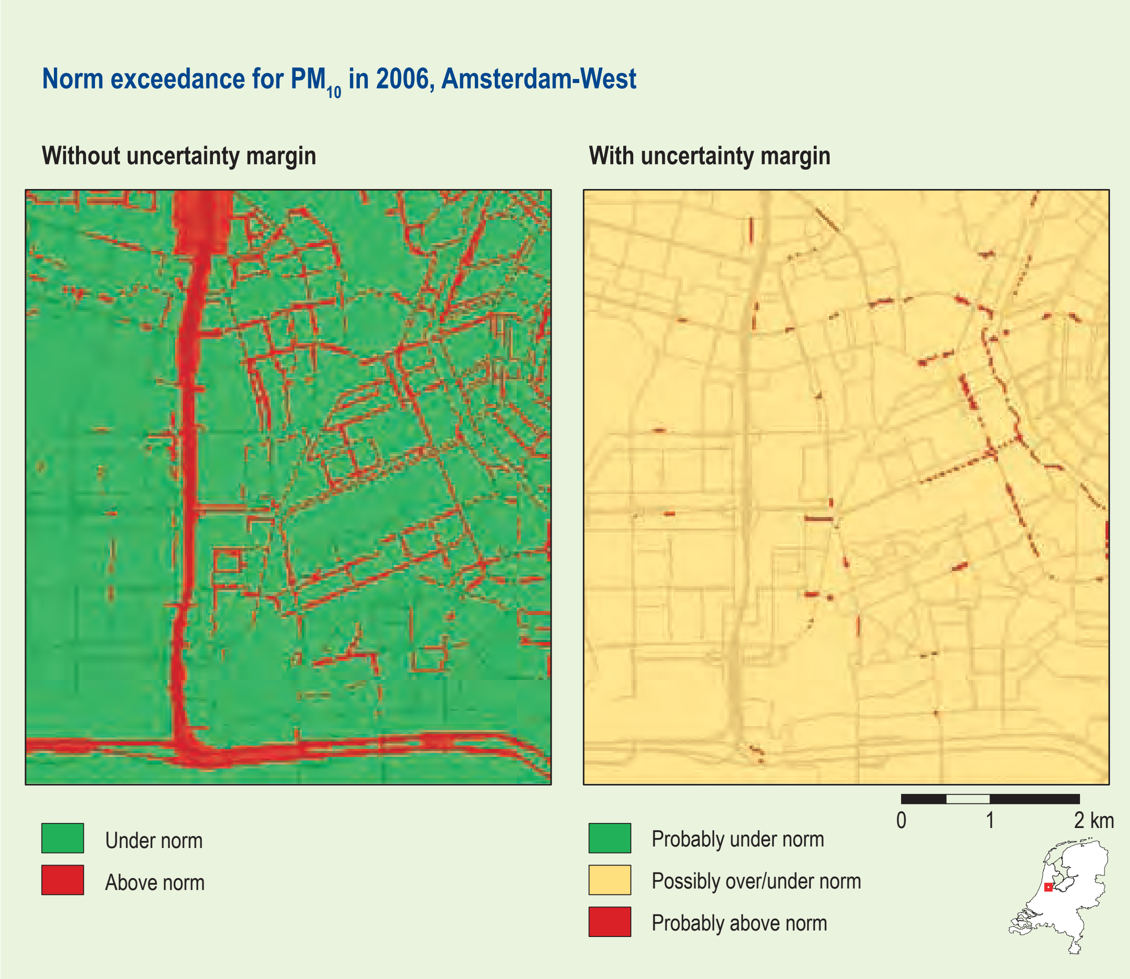

Figure 4.3 Illustration of norm exceedences around the A10 near Amsterdam-West; without an uncertainty margin (left) and with an uncertainty margin (right).

barriers along the A10, while the figure on the right shows that the exceedence situations in sections of streets in the inner city are more certain than those near the A10. Based on this figure, the authorities might consider more generic policy to reduce the concentrations.

4.3 Noise

In the conclusions about environmental noise, the uncertainties are explained in a more limited fashion. This partly has to do with a longer tradition of centrally-established calculation regulations. Consequently, as with air traffic, there is not an immediate need for communication about uncertainties. The limited communication about uncertainties in the noise analyses also has to do with the complexity surrounding noise modelling at the national scale. Due to this complexity, the uncertainties are not yet well understood.

One example where there is attention to uncertainty in the policy evaluation is the analysis of the development of noise hotspots near motorways and railroads. (See text below Figure 2.2 (Section 2.2.3)) This analysis is presented with a statement of the possible disappointing acoustic performance of quiet road surfaces and its effect. The possibility of this disappointing performance emerged from a monitoring programme of the PBL.

4.4 Health

Gaps in our knowledge should not prevent us from making policy decisions. But how can we define robust policy strategies given the various uncertainties in sources, exposure and causes of health effects? Which uncertainties are the most important? How can we find policy strategies that are robust and minimize financial, health and environmental risks?

Uncertainties are partly caused by a lack of data – or of reliable data – which can be solved by better measurements. With statistical techniques (such as error propagation) we can estimate the likelihood that an abatement measure will reduce human health risks based on our estimates of the uncertainty margins in the input data. It is also possible to estimate the cost of the remaining uncertainty for a particular decision. This estimate is called value of information, and it can be calculated within a cost-benefit analysis. Value of information helps in identifying the most critical uncertainties and different types of measures among all policy options. 'No-regret' measures are cost-effective despite remaining uncertainties. Potential 'regret' measures show low cost-effectiveness if one or more assumptions actually deviate from the expectation. Methodological and statistical uncertainties are frequently presented in PBL reports and tables and figures therein.

Besides uncertainties or incompleteness of data, the question of "what precisely causes the health problem?" covers the major uncertainty in some environmental issues. This type of uncertainty is called "conceptual uncertainty" or "incertitude". Two critical incertitudes related to uncertainty about the causality of health effects of air pollution are concerned with the different potencies of various constituents of the fine particulate mass and the level of public health protection that should be applied. This is because many of the considered measures are rather specific in reducing only one or a few constituents of the total fine particle mass, and the question still remains whether this will reduce health effects and to what extent the population should be protected in case of an apparent lack of a threshold.

To help environmental assessors better deal with the science–policy–society interface, in particular to deal with uncertainty and framing of policy problems, the PBL, together with Utrecht University, developed a 'Guidance for Uncertainty Assessment and Communication' (Petersen et al., 2005). In this document, six parts of environmental assessments are distinguished which merit separate attention: (1) problem framing; (2) involvement of stakeholders; (3) selection of indicators; (4) appraisal of the knowledge base; (5) mapping and assessment of relevant uncertainties; (6) reporting of the uncertainty information. The use of this guidance procedure within the PBL in dealing with these types of uncertainties is beginning to develop, but will not be dealt with in this audit report.

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Appendix I Noise propagation

In EMPARA, noise propagation (transmission) – in accordance with the calculation models – uses a geometric dispersion term D_d, air attenuation D_{air}, and correction terms for the absorption by the surface $D_{surface}$ and for weather conditions D_{meteo}. These four components are described in empirical formulas and can be added together.

$$T = D_d + D_{air} + D_{surface} + D_{meteo}$$

The separate terms are made uniform based on the definitions – which is contrary to the calculation and measurement regulations – but the parameterization of the formulas for road noise and railroad noise differs for D_{air} en $D_{surface}$ because a different standard spectrum is used for these types of noise.

The geometric dispersion of a line source is shown by the following:

$$D_d = -10 \cdot \log \frac{\theta}{\pi \cdot d}$$

where θ is the angle of sight with which the line source is viewed from the observation point, and d is the perpendicular projection distance from the observation point to the line source.

$$D_{meteo} = 3{,}5 \; 1 - e^{-0{,}04 \frac{r}{h_r + h_{road} + 0{,}5} - 2}$$

where h_r is the elevation of the observation point above ground level, and h_{road} is the elevation of the road above ground level.

For *road traffic noise,* the air attenuation and ground attenuation are calculated as follows:

$$D_{air,road} = 0{,}008 \cdot r^{0{,}85}$$

where r is the distance to the driving line.

$$D_{surface,road} = 2(B-1) + B\left[4(1-e^{-0{,}028r})(e^{-0{,}065h_r} + 0{,}614e^{-0{,}65h_{weg}})\right] + 3(B-1)(1-e^{\frac{-0{,}0065r}{h_r + h_{road} + 0{,}4}}$$

The first term in $D_{surface,\ road}$ is different than the definition in the calculation regulations. The emission factor has been adapted accordingly. The final term is an additional term for the area in the middle.

For *rail traffic noise*, the air attenuation and ground attenuation are calculated as follows:

$$D_{air,rail} = 0{,}011 \cdot r^{0{,}86}$$

and

$$D_{surface,rail} = 1{,}6B - 1{,}8 + 3\sqrt{B}(1 - e^{-0{,}02r})(e^{-0{,}9h_r} + 0{,}86e^{-0{,}54h_{track}}) + 3(B-1)(1 - e^{\frac{-0{,}0065r}{h_r + h_{track} + 0{,}4}})$$

Appendix 2 Noise – Transmission attenuation due to barriers

The formulas that are used to determine attenuation resulting from noise barriers near a source (line source or otherwise) are as follows:

$$h_b = h_{weg} + 0.75$$

$$h_e(x_w) = H_s - \quad h_b + \frac{x_s}{x_w} \cdot (h_w - h_b) + x_s \cdot \quad \frac{x_w - x_s}{16 \cdot x_w}$$

$$z(x_w) = \quad \frac{1}{x_s} + \frac{1}{(x_w - x_s)} \quad \cdot \frac{h_e^{\,2}(x_w)}{2}$$

$$D_{scherm}(x_w) = 10 \cdot \log \; 3 + A \frac{z(x_w)}{1 + B \cdot x_w} \text{ if } h_e(x_w) > 0$$

or

$$D_{scherm}(x_w) = 10 \cdot \log(3) \cdot \exp(-5 \cdot \sqrt{z(x_w)}) \text{ if } h_e(x_w) \leq 0$$

Where x_s and x_w are the distance from the source to the barrier and the distance to the observer (receptor), respectively. h_b and h_w are the elevations of the source and observer, h_{weg} is the elevation of the road. H_s is the height of the barrier. All distances and elevations are expressed in metres; elevations are shown with respect to the reference level (average ground level near the source).

The variable $h_e(x_w)$ indicates the effective barrier height (i.e., the part of the barrier that projects above the deflected sound beam) as a function of the source-receptor distance. $z(x_w)$ is the acoustic path difference, which determines the noise-attenuating effect D_{scherm} of the barrier. To account for possible singularities for receptor points immediately behind the barrier, the term $(x_w - x_s)$ in the denominator of $z(x_w)$ is replaced by $(x_w - x_s + .001)$ during implementation of NOISTOOL2/¾. The coefficient A is set equal to 80 and B to 5.10^{-4}.

The above formula is exemplary for infinitely long, absorbing barriers parallel to a straight road, but is used in NOISTOOL2/¾ for calculating the transmission attenuation for *partial line segments* with barriers, where the source-receptor direction is *not necessarily perpendicular* to the line source direction. All distances are measured along the bisector between the line segment midpoint and the observation point. This is shown graphically in Figure A.1.

Geometry for determining Dhuis

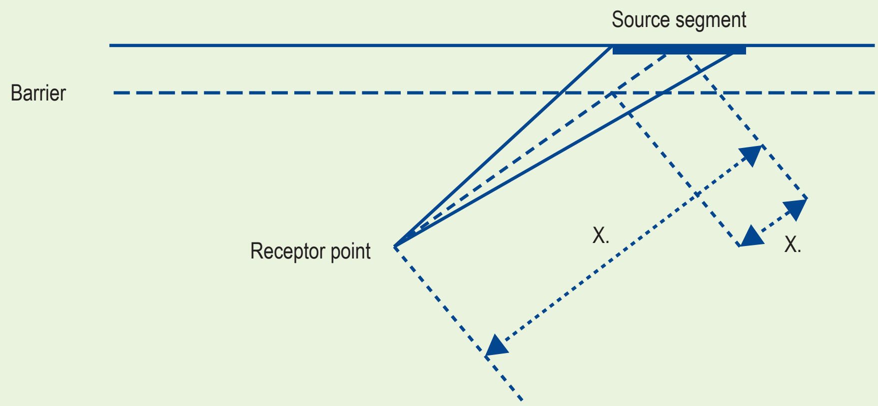

Figure A.1 Geometry for determining D_{huis}, showing the definition of distance.

Appendix 3 Noise – Object attenuation in the urban area

In the above object attenuation model, a distinction is made between a *penetrating beam*, which after attenuation (via τ_0), undergoes additional object attenuation, and a *barrier beam*, which travels from the top of the first line of buildings to the immission point without additional object attenuation (effect D_s). $D_{object,jj}$ results from a combination of both components in this 2-beam model. The variable C_o determines the attenuation of the penetrating beam and is set to 450 for urban built-up areas and to zero for non-urban built-up areas.

$$h_b = h_{weg} + 0.75\,;\; \tau_0 = \frac{lv_0}{lv_0 + C_0}$$

$$D_t(x_w) = 4.3 \cdot \frac{(x_w - x_b)}{lv_1} - 10 \cdot \log\left[1 + 5(1-\alpha_1)\cdot\left[\frac{(x_w - x_b)}{lv_1}\right]^{1.5} \cdot \exp\left(\left[1 - \sqrt{\alpha_1 \cdot (2-\alpha_1)}\right]\cdot\frac{(x_w - x_b)}{lv_1}\right)\right] - 10 \cdot \log(\tau_0)$$

$$h_e(x_w) = H_{beb} - \;\; h_b + \frac{x_b}{x_w}\cdot(h_w - h_b) + x_b \cdot \;\; \frac{x_w - x_b}{16 \cdot x_w}$$

$$z(x_w) = \;\; \frac{1}{x_b} + \frac{1}{(x_w - x_b + 0.001)} \;\; \cdot \frac{h_e^{\,2}(x_w)}{2}$$

$$D_s(x_w) = 10 \cdot \log \;\; 3 + A\frac{z(x_w)}{1 + B \cdot x_w} \text{ if } h_e(x_w) > 0$$

or

$$D_s(x_w) = 10 \cdot \log(3) \cdot \exp(-5 \cdot \sqrt{z(x_w)}) \text{ if } h_e(x_w) \le 0$$

$$D_{object,jj} = -10 \cdot \log\left(10^{-0.1 D_t(x_w)} + \left(1 - 10^{-0.1 D_t(x_w)}\right)\cdot 10^{-0.1 D_s(x_w)}\right) \text{if } x_w > x_b$$

and otherwise

$$D_{object,jj} = 0$$

In the formula, x_b and x_w represent the distance from the source to the first line of buildings and the observer, and h_b and h_w the elevation of the source and observer, while H_{beb} is the building height, h_{weg} is the elevation of the road, lv_0 is the free path length in the first line of buildings, and lv_1 is the free path length behind the first line of buildings. All lengths, distances and elevations are expressed in meters; the elevations are given with respect to the reference surface, i.e. the average ground level near the source. α_1 indicates the average object absorption behind the first line of buildings. A is set equal to 80 and B to 5.10^{-4}.

Appendix 4 Health Impact – Indicator specification

In relation to its aim, application, users and potential target group, the scope of the health impact assessment can be framed with support of a causal environmental health network (see example in Figure A.2). The DPSEEA (Driving Forces - Pressures - State - Exposure - Effects - Actions) model is useful in designing a system of EH indicators within the decision-making context. One or more of the model components can be developed and used for reporting and communicating the most essential aspects of a specific assessment, and accordingly serve as indicator.

Indicator development
A list of attributes is provided for the development of indicators. This structured format allows for clear-case indicators that can be re-used or updated over time.

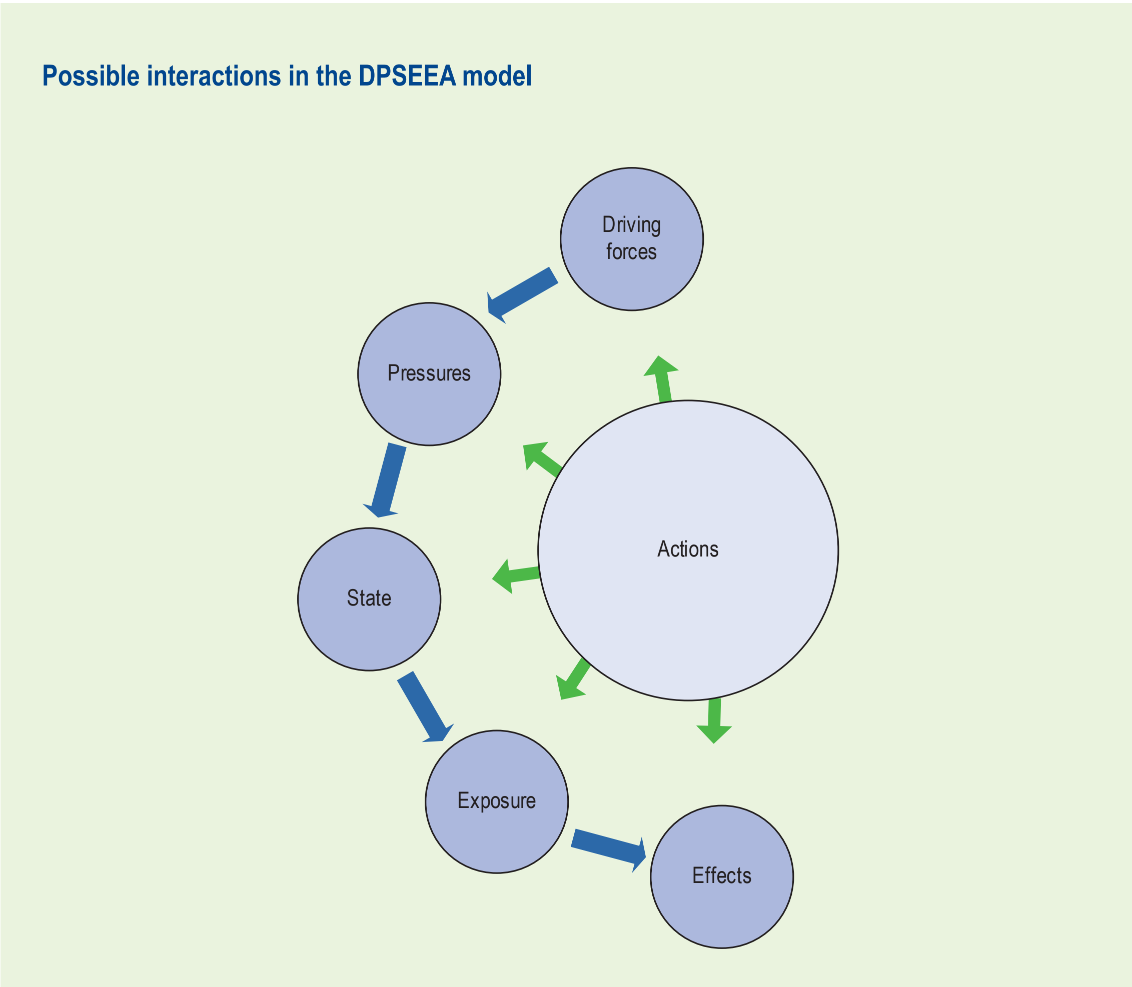

Figure A.2 The DPSEEA model demonstrates the possibilities for action throughout the interaction chain, distinguishing five components: driving forces, pressures, state, exposure and effects of health-environment interlinks.. Driving forces refer to societal factors that push environmental processes. Environmental pressures result in modification of the environmental state. Exposure to enviornmental hazards may result in adverse health effects. (WHO, 2005).

1. Name – What is the name of the indicator?
2. Scope - What question does the indicator attempt to answer?
3. Unit – What is the unit of measurement?
4. Definition - How can you derive or calculate the answer?
5. Result and discussion - What is the answer to the question defined in the scope and why is this answer given?

The name attribute is the identifier of the indicator. The indicator names should be chosen so that they are descriptive, unambiguous and not easily confused with other indicators.

The scope attribute defines the boundaries of the indicator - what does it describe and what does it not describe? The boundaries, for example, can be spatial, temporal or abstract.

The unit attribute describes the units with which the result is presented. The units of interconnected indicators must be coherent with each other in a causal network description.

The definition attribute describes how the result of the variable is derived. It consists of sub-attributes to describe the causal relations, data used to estimate the result and the mathematical formula to calculate the result. In addition, alternative, identified ways to derive the variable result can be described in the definition attribute as a reference.

The result attribute describes the result value of the variable. The result value can be represented in various forms, but is preferably quantitative (i.e. numerical). (Tuomisto and Pohjola, 2007)

Indicator reporting
Assessment reporting is most informative to the users if indicators are presented in a structured way. According to the European Environment and Health Information System indicators must be described including the following elements (WHO, 2008):
- Indicator name, definition
- Key message & rationale
- 1-2 charts (with description)
- Health and environment context
- Policy relevance and context
- Assessment (including HIA)
- Meta-data (data-about-data)
- References

Most importantly, a clear message should be conveyed to the intended user of the assessment outcome, which is defined in relationship to the assessment purpose: the analysis or evaluation of a policy.

Appendix 5 Health Impact - Policy targets

Performance targets

Distance to target		
Long Term Exposure (annual mean)	$PM_{2.5}$	10 µg/m³, guideline (WHO AQG 2005) 25 µg/m³, limit value to be met in 2015 (EU proposal) 20 µg/m³, Exposure Concentration Obligation (ECO) in urban background, limit value to be met in 2015 (EU proposal) 15% reduction of Annual Exposure Index (the annual average background concentration indexed at 2010), Exposure Reduction Target (ERT) value to be met in 2020 (EU proposal)
	PM_{10}	20 µg/m³, guideline (WHO AQG 2005) 40 µg/m³, limit value (EU 1999/30/EG; Wm Hst 5)
	Ozone	–
	NO_2	40 µg/m³, guideline (WHO AQG 2005) 40 µg/m³, limit value (EU 1999/30/EG; Wm Hst 5)
	Noise	See section 4.2
Short Term Exposure (maximum number of days per calendar year above level)	$PM_{2.5}$	0 days with 24-hour mean above 25 µg/m³, guideline (WHO AQG 2005)
	PM_{10}	0 days with 24-hour mean above 50 µg/m³, guideline (WHO AQG 2005) 35 days with 24-hour mean above 50 µg/m³, limit value (EU 1999/30/EG; Wm Hst 5)
	Ozone	0 days with 8-hour mean above 100 µg/m³, guideline (WHO AQG 2005) 25 days with 8-hour mean above 120 µg/m³ on average in 3 years, target value to be met in 2010 (EU 2002/3/EC; Wm Hst 5) 0 days with 8-hour mean above 120 µg/m³ on average in 3 years, target value to be met in 2020 (EU 2002/3/EC)
	NO_2	0 days with 1-hour mean above 200 µg/m³, guideline (WHO AQG 2005) 18 days with 1-hour mean above 200 µg/m³, limit value (EU 1999/30/EG; Wm Hst 5)
	Noise	–
TSAP[1)] goal for 2020	$PM_{2.5}$	5,6 (TSAP) / 5,8 (NEC[2)]) million years of Life Lost (YOLL) (47% improvement compared to YOLL 2000)
	PM_{10}	–
	Ozone	323 (NEC) / 374 (TSAP) number of cases of premature deaths (10% improvement compared to YOLL 2000)
	NO_2	–
	Noise	–

[1] TSAP is an abbreviation for Thematic Strategy on Air Pollution
[2] NEC is an abbreviation for National Emission Ceiling

Appendix 6 Health impact - Calculations according to different models

YLL according to RAINS

RAINS calculates YLL from exposures to $PM_{2.5}$. It uses life tables to calculate the survival function. UN population data are used. The survival function (l_t) indicates the percentage of a cohort alive after time *t* has elapsed. It is an exponential function of the sum of the mortality rates. Because the relative risk function from Pope et al. (2002) applies only to those cohorts that are at least 30 years old, younger cohorts were excluded from the analysis. Accordingly, for an age cohort aged *c* at start *s, lc(t)* is:

Survival function $l_c(t) = e^{-\sum_{z=c}^{t} \neg_{z,z-c+s}}$

The remaining life expectancy e_c for a cohort aged *c* is the integral from *c* to *w1* over *lc(t):*

$$e_c = \int_c^{w_1} l_c(t)dt$$

Life expectancy where *w1* is the maximum age considered (95 years in this case).
The absolute change in life expectancy per person of a cohort c in year s is

$$e_c = (\beta \cdot PM)\int_c^{w_1} l_c(t)\ln l_c(t)dt = (\beta \cdot PM)H_c$$

where

$$H_c = \int_c^{w_1} l_c(t)\ln l_c(t)dt$$

The change in life years for all persons of one cohort in grid cell *x,y* is obtained by multiplying the above equation by the size of the cohort *Pc/x,y* and the length of the time interval for which demographic and mortality data are given. This leads to the change in life years lived for cohort *c* in grid cell *x,y*. Because cohort data were obtained with reference to the aggregate national level, cohort size in a grid cell was calculated by weighting total population in a grid cell with the relative share of the given cohort in the national population.

For all cohorts in a grid cell x,y the change of life years is expressed as the sum of the change in life years for the cohorts. Dividing this by total population at least of age 30 in grid cell x,y leads to the average change in life expectancy in grid cell x,y. In order to calculate the average change in life expectancy for a country, the change in life years in all grid cells of a country divided by total population is computed. The methodology uses *linear approximations* for the hazard rate, i.e. of the relative risk and for calculating absolute changes in life expectancy accordingly. Uncertainties were dealt with by doing a partial sensitivity analysis using the upper and lower bounds of the 95% confidence interval of the relative risk (from Pope et al., 2002). Therefore, loss of life expectancy (LLE) was calculated with RR 1.02 and 1.11, resulting in 11.8 months lost per person in 2000. The total years of life lost in the Netherlands for 2000 is estimated at 10.55 million by IIASA. This is the loss of life years that the currently alive population will experience over its lifetime. (Amann, 2005)

YLL according to the IOMLIFET model

The institute of Occupational Medicine (IOM, Edinburgh) also uses life tables to calculate years of life lost due to exposure to environmental pollutants. Changes in mortality rates imply changes in survival distributions, and these can be estimated using standard life-table calculations. Impacts of air pollution can thus be reflected in the survival rates and years of life lost accordingly. The force of mortality or hazard is calculated first, and the probability of surviving is determined using mid-year population data and the number of deaths. The hazard is defined as the instantaneous probability of death at a particular time. The relationship between this quantity and the probability of surviving a period of time is the basis of standard life table methods of describing mortality patterns. The average hazard rate for each year is estimated from observed data as number of deaths d divided by the mid-year population m, as shown in the following formula:

$$H = \frac{d}{m}$$

The probability of surviving is then:

$$S = \frac{m - \frac{1}{2}d}{m + \frac{1}{2}d}$$

Of course, hazard rates increase with aging. The probability of surviving over a number of one-year periods is calculated by multiplying the individual one-year survival probabilities. The expected average length of life from birth can then be calculated by summing the life years over all periods. Exposure to air pollution increases the hazard rate. Using the relative risk for PM_{10} mentioned before (1.043 per 10 μg/m³) survival curves can be compared, as well as years of life lost. (Miller, 2006).

To calculate the impact of PM_{10} concentrations in 2000 in the Netherlands, compared to no pollution (no threshold), population data from Statistics Netherlands was used, as well as PM_{10} concentrations from Knol et al. (2005). Exposure to PM_{10} changes the hazard rates, and this allows the change in life expectancy and change in number of deaths to be calculated. Years of life lost were only calculated in 2000 for the population above 30 years of age. Using 1-year age groups and a concentration of 31.35 μg/m³ PM_{10}, the population in 2000 loses around 175,000 life years. This population will experience 10.16 million lost life years over its lifetime.

YLL according to WHO AirQ model

The Air Quality Health Impact Assessment Tool (AirQ) is specialized software that enables the user to assess the potential impact on human health of exposure to a given air pollutant in a defined urban area during a certain time period. The impact of a pollutant on human health is considered with respect to mortality. Quantification of the health impact for the exposure to the air pollutant is based on the population-attributable risk proportion.

$$AP = \frac{\sum (RR(c)-1)p(c)}{\sum RR(c)p(c)}$$

where:
RR(c) is the relative risk for the health outcome in category c of exposure
p(c) is the proportion of the population in category c of exposure

The model allows country-specific data to be used, and this was done for the Netherlands for the year 2000. Impacts could be calculated using air quality data or specific population and mortality data. PM_{10} levels for the year 2000 were collected from the National Measurement Network Air Quality (LML), taking into account the criteria for validity of stations. To obtain one-hour average values from data with a smaller averaging time, at least 75% of valid data should be used; to obtain 8-hour "moving" average values from hourly measures, valid measurements have been performed for at least 18 hours (75%); to obtain 24-hour average values, data should be available for at least 50% of the time, and to obtain seasonal and annual average values, at least 50% of valid data for the reported period should be used. Although PM_{10} concentrations for all the stations available was collected and separated according to the categories 'urban station', 'pre-urban station', 'regional station' and 'street station', only one was selected because it was too laborious to calculate all the input required, and this was only used as an example. For this station, an urban station (number 418, in Rotterdam) the annual mean, maximum and 98^{th} percentile was calculated for the year 2000. This was also done for the summer and winter. Data had to filled in for 342 days for the whole year. This comprised 163 days in the winter (January through March and October through December) and 179 days in the summer (April through September). In addition, the number of days with certain PM levels had to be filled in. The categories distinguished were: less than 10, 10 to 19, 20 to 29, 30- 39, etc. The unit was µg/m^3 PM_{10}. Most of the days, 127 in total, had PM_{10} levels between 20 and 29. The annual mean was 34 µg/m^3 PM_{10}. The population exposed had to be filled in, which was the total number of people above 30 years of age in the Netherlands in 2000.

Next the health endpoint of PM_{10} had to be selected, which was total mortality in this case. Baseline incidence had to be filled in (mortality rate) and the relative risk was applied. This resulted in an estimated number of excess cases of about 14,000.

When using life tables, the PM_{10} data was also needed except for the categorized information (number of days with certain levels). The number of deaths and mid-year population by age (5-year age categories) for the year 2000 had to be filled in. The pollutant concentration for which the RR are scientifically valid were needed, as well as the reference level. After filling in the RR, years of life lost could be calculated. Years of life lost were approximately 190,000. It is also possible to make scenarios, although this was not done.

A summary of the outcomes using the different health impact assessment methods is given in Table 4.1. It should be stressed that 'deaths' and 'YLL' as a health outcome can not be converted to each other. Regarding the number of deaths, it is unclear how many life years are lost since it not known when (which age) they die; when determining the YLL it is unclear how many people this includes and how many life years they lost individually.

Table 4.1 Mortality associated with long term exposure to PM_{10}

Duration[1]		Indicator[2]	RIVM	RAINS	CAFE	IOM-LIFET	WHO AirQ
1 yr		# deaths	18.1 thousand	–	15.5 thousand	15.6 thousand	14 thousand
1 yr		YLL_1	178 thousand	–	184 thousand	175 thousand	190 thousand
∞		YLL	–	10.55 million	–	10.16 million	–
∞		LLE	–	0.983	–	1.023	–
PM		*• fraction:*	PM_{10} total	$PM_{2.5}$ human	$PM_{2.5}$ human	PM_{10} total	PM_{10} total
		• level:	31.4 μg/m³	19.4 μg/m³	19.4 μg/m³	31.4 μg/m³	34 μg/m³
		• data:	meas./model	model	model	meas./model	measurement
			GCN 2000	RAINS 2000	RAINS 2000	GCN 2000	LML 2000
Population		*• cohorts:*	no breakdown	5 yr	..	1 yr	..
		• group:	total	30+	total / 30+	30+	30+
		• data:	CBS	UN	UN	CBS	CBS
Method		*• models:*	PAR only			Life Table	
		• period:	2000 [1 yr]	2000-2100	2000-2100	2000-2100	
		•timesteps:	1 yr	5 yr		1 yr	
Population:		*• cohorts:*	no breakdown	5 yr	..	1 yr	..
		• group:	total	30+	total / 30+	30+	30+
		• data:	CBS	UN	UN	CBS	CBS
Method:	*Other differences/ deviations:*		LE estimated			Definition S	mid-year pop
	Model period:		2000	2000-2100		mid-year pop	
	Reference:		Knol (2005)	IIASA (2005)	AEAT (2005)	Miller (2003)	WHO (2004)

[1)] duration of intervention: 1 year or lifetime (∞)

[2] for the total population in the Netherlands (16.3 mln inhabitants)

[3] based on an estimated number of years of life lost per death

Appendix 7 Health impact - Input data for calculations

References with parameters health effects have been divided into 5 classes. Parameters that are more applicable to the situation in the Netherlands are preferred. The parameters have been taken from the following publications, in declining order of preference:

A. MGO (Knol & Staatsen 2005)
B. Dissertation De Hollander (2004)
C. EU-project CAFE HIA/CBA methods (2005)
D. AirQ model 2.2.3 (WHO) (2004)
E. Additional (IVM report, WHO-Europe)

Relative risks (RR) values for air pollution are given per 10 μg/m³. The distribution is presented as σ. BR presents baseline rates.

Duration and severity of the effect are listed in the rightmost column with clear reference to the source.

Health endpoints		**Parameters**							
Effect	**Symptom**	**RR**	**σ**	**ref**	**BR**	**ref**	**duration**	**severity**	**ref**
PM_{10} long-term exposure									
mortality	Overall	1.043	0.009	A	0.01822	A	10	1	A
	Lung cancer	1.02	0.542	B	0.00057	A	13	1	B
	Cardiopulmonary	1.043	0.027	B	0.0032	B	8.2	1	B
	infant (1st 2 months life)	1.04	0.013	C	0.00033	E	77.9	1	E
chronic	resp. symptoms children	1.087	0.041	B	0.057	B	1	0.17	B
health	chronic bronchitis	1.306	0.094	D	0.096	E	1	0.03	E
symptoms	bronchitis adults	1.025	0.007	B	0.018	B	1	0.31	B
	chronic cough child 6-15	1.61	0.11	E	0.068	E	1	0.03	E
$PM_{2.5}$ long-term exposure									
mortality	Overall	1.06	0.023	A	0.00837	B			
	Cardio-respiratory> 30	1.06	0.01	E	0.00417	B	7.7	1	E
	Lung cancer > 30	1.08	0.01	E	0.0006	B	8.1	1	E
morbidity	chronic bronchitis	1.141	0.078	C	0.076	C	8.4	0.32	E
PM_{10} short-term exposure									
mortality	Total	1.0036	5E-04	A	2.4E-05	B	0.25	1	A
	post-neonatal mortality	1.04	0.013	E	0.00011	E	76.3	1	E
	cardiovascular mortality	1.0025	8E-04	A	7.1E-06	B	0.25	1	A
	respiratory mortality	1.0114	0.002	A	0.00088	B	0.25	1	A
	resp. post-neonatal mortality	1.2	0.077	E	6.1E-07	A	76.3	1	A
	> 65	1.0073	0	A	0.00813	A	0.04	0.65	E
	COPD	1.0106	0.002	A	0.00039	A	0.25	1	A
	Pneumonia	1.0121	0.003	B	6.5E-07	A	0.25	0.77	B
	sudden infant death syndrome	1.12	0.026	E	1.2E-06	A	76.3	1	A

Health endpoints					Parameters				
Effect	**Symptom**	**RR**		**ref**	**BR**	**ref**	**duration**	**severity**	**ref**
HA	cardiovascular disease	1.0032	6E-04	A	0.00609	A	0.04	0.71	A
	Respiratory	1.0047	0.001	A	0.00377	A	0.04	0.64	A
	Pneumonia	1.0069	0	E	0.00135	A	0.04	0.64	E
	COPD	1.0084	0.002	A	0.00093	B	0.04	0.53	A
ER visits	Respiratory	1.015	0.005	B	0.022	B	0.03	0.51	B
aggr asthma	asthmatic attacks	1.044	0.542	B	0.00023	B	0.01	0.22	B
	use of bronchodilators	1.07	0.031	B	0.00059	B	0.01	0.22	B
	incr bronch use child 5-14	1.005	0.012	C	0.1	C	0.02	0.05	C
	incr bronchodilator adults	1.01	0.01	C	0.0045	C			
acute	asthma attack children	1.051	0.002	D			0.01	0.22	D
gezondheids-klachten	asthma attack adults	1.004	0.002	D	0.094	E	0	0.36	E
	acute bronchitis < 15	1.31	0.094	D	0.096	E	1	0.03	E
aggravation of respiratory	lower respiratory tract	1.038	0.017	B	0.038	B	0.04	0.21	B
symptoms	LRS child 6-11 incl cough	1.04	0.01	C	0.015	C			
	lower resp tract child 6-11	1.004	8E-04	C	0.02	C	0.04	0.21	E
	upper resp tract	1.02	0.521	B	0.19	B	0.02	0.05	B
	cough children 6-11	1.004	0.001	C	0.013	C	0.04	0.21	E
	cough symptoms adults	1.043	0.02	C					
LRS adults	Total	1.017	0.008	C	0.04223	A			
	Wheeze	1.059	0.032	C	0.151	C	0.01	0.22	E
	shortness of breath	1.032	0.014	C	0.457	C			
URS&LRS child	children 7-13	1.029	0.005	E	0.15068	E	0	0.04	E
med days	bronchitis, children	1.02	0.012	E	0.06027	E	0	0.23	E
GP consult	asthma 0-14 warm sea.	1.025	0.013	C	0.0471	C	0.01	0.22	E
	asthma 15-64 warm sea	1.031	0.01	C	0.0165	C			
	asthma > 65 warm sea.	1.063	0.023	C	0.0151	C			
	upper resp di < 15	1.007	0.514	C	0.574	C			
	upper resp di 15-64	1.018	0.005	C	0.18	C			
	Upper resp dis > 65	1.033	0.008	C	0.141	C			
$PM_{2,5}$ short-term exposure									
mortality	Total	1.015	0.002	D	0.00837	B			
RAD	18-64	1.0048	3E-04	C	0.05205	C	0.01	0	E
WLD	15-64	1.0046	4E-04	C	0.01233	C	0.02	0	A
MRAD	18-64	1.0074	7E-04	C	0.02137	C			
Ozone short-term exposure									
mortality	total	1.0026	6E-04	A	0.00886	A	0.25	1	A
	cardiovascular	1.0021	1E-03	A	0.00266	A	0.25	1	A
	Respiratory	1.008	0.002	B	0.00088	A	0.25	0.79	B
	Pneumonia	1.0116	0.003	B	0.00034	A	0.25	0.77	B
HA	respiratory disease	1.0042	8E-04	B	0.00927	A	0.04	0.64	B
	respiratory disease < 14	0.999	0.006	E	0.02073	A			
	respiratory disease 15-64	1.001	0.005	C	0.00526	A	0.04	0.65	E
	respiratory disease >65	1.005	0.004	C	0.01506	A	0.04	0.65	E
	cardiovascular	1.007	0.002	E	0.01417	A	0.04	0.71	E

Health endpoints		Parameters							
Effect	**Symptom**	**RR**		**ref**	**BR**	**ref**	**duration**	**severity**	**ref**
	asthma	1.146	0.076	C	0.00034	A			
	asthma < 15	1	0.002	D	0.00088	A			
	asthma 15-64	1.007	0.007	D	0.0002	A			
	COPD	1.0086	0.002	D	0.00093	A			
gp consults	allergic rhinitis 0-14	1.082	0.017	C	0.037	C			
	allergic rhinitis 15-64	1.055	-0.01	C	0.029	C			
RADs	minor	1.0148	0.005	C	0.02137	C	0	0.01	E
RADs		1.0117	0.003	E			0	0.01	E
ER visis	respiratory disease	1.0058	0.002	B	0.035	B	0.03	0.51	B
incr use of bronchilators	in children 7-15 with asthma	1.41	0.214	C	0.01	C	0.01	0	E
	in adults with asthma	1.009	0.006	C	0.045	C			
aggr of asthma	extra attacks	1.0079	0.003	E			0	0.36	E
symptom days	15-64	1.004	0.002	E			0	0.04	E
medication use	symptomic children	1.41	0.214	E					
cough	in children with chronic resp. symptoms	1.04	0.033	C	0.08658	C	0.03	0	C
	children general pop.	1.05	0.033	C	0.1445	E			
	in adults with symptoms	1.05	0.077	E					
breathing problems	lower resp. symptoms children 7-11	1.03	0.059	C	0.015	C			

Assessment of local environment and health and policy evaluation

In June 2008, the Quality of the Local Environment Team (LOK) of the Netherlands Environmental Assessment Agency (PBL) was submitted to an international scientific audit. Subject of the audit were the methods used by the team for the evaluation of government policy, concerning the quality of the physical local environment and the consequences thereof for human health and well-being. This concerns the dossiers air traffic, noise, air quality, external safety and health in relation to environmental quality.

This report has been compiled to provide the audit committee with the necessary information to conduct the audit.

The present report extensively addresses the methods and data that are important to these dossiers and the uncertainties that play a role in them. A separate chapter describes how uncertainties are dealt with in the policy evaluation.

Besides making a scientific evaluation of the work, the audit committee has also been requested to evaluate whether the scientific research sufficiently links up with the policy evaluations. The report, therefore, briefly describes the relevant policies in the various dossiers, with examples of policy evaluations.

Netherlands Environmental Assessment Agency, October 2008