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Air quality mapping of NO_2 with the use of passive samplers

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Preface

This report has been written for, and on the request of, the Norwegian State Pollution Control authority (SFT), contract number 3008065. It presents an overview, assessment and recommendation on methods for making air quality maps of NO_2, based principally on the use of passive samplers.

Contents

Summary

The European Commission requires the reporting of air quality for a number of legislated air pollutants, including Nitrogen Dioxide (NO_2). Such reporting is required in all the territories of the Member States. There are a number of methods available for providing such information, generally based on monitoring data, but also other methods such as 'indicative measurements' or modelling may also be used. The major drawback with monitoring is that it assesses the air quality at only one point in space, whereas the assessment is actually required everywhere.

To assist in producing spatial maps of NO_2 concentrations this report details methodology for creating such maps in a cost effective way, with the use of passive samplers. It provides an overview of passive sampling methods and their costs and quality. It then provides a statistical assessment of the uncertainty in converting passive sampler measurements to annual mean and percentile concentrations, based on an analysis of measurement data available in Nordic countries. It then reviews the current state-of-the-art in air quality mapping using both passive samplers and other data sources. Finally it describes and recommends different mapping methodologies.

The best mapping method is to combine a well planned passive sampling monitoring network with dispersion modelling and other spatially distributed data. In this way a high level of detail can be obtain in such maps, covering both the local hotspot and urban background scales. It is not highly recommended to use passive sampling alone for detailed mapping though such a method can be used for mapping urban background levels.

It is also presented that passive sampling does not fulfil the air quality directive quality objectives for 'fixed measurements', but instead falls under the category of 'indicative measurements'. As such passive sampling alone cannot be used for directive assessments when the air quality levels are close to or above the European limit values.

Air quality mapping of NO_2 with the use of passive samplers

1 Introduction

There is a need for all cities within Europe to assess their air quality. This is stipulated in the new directive on air quality (EC, 2008) which came into force in June 2008. This directive is intended to streamline and replace already existing directives. Within this directive limit values for a range of pollutants are provided and assessment must be carried out by all Member states, including Norway, in all their territory. Though particulate matter is generally accepted to be the most problematic of the legislated pollutants in Norway there are still exceedances of the EU limit values of NO_2 in some Norwegian cities.

Assessment can be carried out using monitoring or modelling methods. The establishment of comprehensive automatic monitoring stations, that provide hourly average concentrations of a range of pollutants, is generally an expensive undertaking. In many cases this may not be necessary as the pollutant levels may be so low as to not warrant their use. In addition the investment in a single station provides information at one point in space only. In regard to the air quality directive the assessment must be carried out 'everywhere' within a zone.

Models provide a complete spatial distribution of air quality but the results of models are often seen as less certain than monitoring for a number of reasons. E.g. that the model description is not perfect, that the emission data is not well known or that other input data such as meteorology is not known or not well defined. To ensure some form of quality control of models, validation using monitoring must always be carried out to assess the models performance.

As an alternative to automatic monitoring stations and air quality models it is also possible to use passive sampling techniques to determine air quality (also known as diffusive samplers). Passive samplers are not ventilated and as such do not require pumps and accurate measurements of air flow and intake. They are small and inexpensive and do not need electrical supplies or continuous maintenance. The major drawback with such samplers is that they require integration times, i.e. exposure to ambient air, of roughly a week, dependent on concentration levels. They are also considered to be less accurate than automatic ventilated monitoring stations. Passive samplers have been extensively used for pollutants such as NO_2, O_3, SO_2 and Benzene but their application to pollutants such as PM is still not well technically established.

This report provides background information and methodological descriptions of methods for mapping NO_2 concentrations using passive samplers. Though passive sampling has been used extensively for screening purposes this does not always provide the complete territorial coverage as required by the air quality directive. Maps are also useful visual tools for communicating the results.

1.1 Air quality directives for NO_2

The 2008 directive (EC, 2008) on ambient air quality places a limit value on both the annual mean concentration and the 19'th highest hourly mean concentrations of NO_2. These limit values must be adhered to in all areas within the nationally defined zones where people either live or are present. Excluded from this directive are industrial terrains (which come under another directive) and carriage ways of major roads. The directive is thus valid at all residential addresses, all pedestrian areas, e.g. curb side, as well as parks and recreational areas.

The air quality directive provides information on the required assessment method and quality. The methods used to assess the air quality are dependent on the air quality levels themselves. The directive defines two threshold levels in this regard. The first is the upper threshold level, above which 'fixed measurements' are required that have an uncertainty < 15%. The second is the lower threshold level, above which 'indicative measurements' or modelling in combination with 'fixed measurements' may be used to assess air quality. The 'indicative measurements' and modelling methods have a lower quality objective, requiring an uncertainty of < 25%. Below the lower threshold value other methods, described as 'objective estimates', may be used. Modelling may be used at all levels but can be used exclusively only below the lower threshold level. In addition to the quality objectives of these methods there are also demands on the temporal coverage of the measurements. These limit values, threshold levels and quality objectives are provided in Table 1 - Table 3. The air quality directive thus provides flexibility for the methods applied for the assessment, depending on the level of concentrations.

In addition to the limit values on NO_2 for the protection of human health there are also limit values on NO_X for the protection of vegetation. The critical level for the protection of vegetation is an annual mean NO_X concentration of 30 $\mu g/m^3$. This is in general less problematic than the limit values set for human health since it is only valid in rural areas outside of populated regions.

Table 1: Air quality directive limit values for NO_2.

Air quality directive limit values for NO_2 ($\mu g/m^3$)	Limit value	Upper threshold	Lower threshold	Alert threshold
Annual mean	40	32	26	-
19'th highest hourly mean	200	140	100	400

Table 2: Air quality directive assessment levels for NO_2.

Air quality directive assessment levels for NO_2	Method
Above upper threshold	Fixed measurements supplemented with modelling and/or indicative measurements
Above lower threshold	Combination of fixed measurements, indicative measurements and modelling
Below lower threshold	Modelling and/or objective estimates

Table 3: Air quality directive quality objectives for NO_2.

Air quality directive quality objectives for NO_2	Uncertainty	Temporal coverage
Fixed measurements	< 15 %	> 90 %
Indicative measurements	< 25 %	> 14 % (1 day a week or 8 weeks a year)

1.2 Screening or mapping?

Passive sampling is an often used method for screening purposes. Screening is a preliminary assessment for indicating the level of a particular pollutant. If the level is found to be below a predefined level then usually no further assessment is carried out. However, if the screening indicates higher concentration levels then further assessment, either through a more extensive monitoring network and/or through modelling activities, should be carried out.

When passive samplers are used for screening purposes then usually sites are selected that are seen to be representative for a particular situation. For example measurements may be made at an urban background site (a site not directly influenced by any particular source), at a major highway, in street canyons, along open roads, downwind of industrial sites, in parks or in rural areas. Screening tries to take representative samples to indicate the expected levels of concentrations at similar types of sites. Screening should also be carried out at different times of the year since meteorology can vary significantly from one season to the next. Generally screening should be carried out, at a minimum, during both summer and winter seasons.

The major aim of mapping, in contrast to a screening study, is to provide a spatial representation of air quality for all of the area of interest. Implicit with any mapping approach is the question of resolution. What is the minimum area resolved by the map? What is the spatial representativeness of the monitoring data used? In many mapping applications, when models are not applied, interpolation may be used to fill in the areas where measurements or other information are not available. The different aims of screening and mapping will thus lead to the use of different methodologies and to different network designs, when monitoring is to be used.

In both applications the question of uncertainty arises. In the case of screening the instrumental error is important but also the choice of screening sites contributes to the uncertainty of the assessment. Are there hotspot areas not covered by the screening? Are the background levels sufficiently well represented? The question of uncertainty in mapping is most important when interpolation is used to create maps. A map can be made from any set of data, either objectively or subjectively, but the uncertainty of the map will be dependent on the methodology used and on the amount of data (and its quality) available. It is therefore important to indicate uncertainty whenever maps are to be made.

2 Technical aspects concerning passive sampling of NO_2

2.1 General Information

Several passive samplers for measuring NO_2 are available on the market and a list of some of these can be found in the Table 4. All types (badge, palmers and radial tubes) are based on the property of molecular diffusion of gasses and hence the term passive, or diffusive, samplers as they need no pump or electricity. The characteristics of each passive sampler differs slightly between instruments so direct contact with each producer or retailer is recommended if a more specific and detailed description of the samplers is required. Chang et al. (2008) provides a good and extensive review of the development, characteristics and application of passive samplers for NO_2.

The samplers give an integrated average, with the averaging time being the same as the exposure time, typically between one day to a month. This means that an assessment of percentiles, e.g. the 19th hourly limit value required in the European air quality directive, see Table 1, is not directly possible. It is possible to use statistical analysis of hourly NO_2 monitoring data to provide an estimate of percentile values, based on annual mean concentrations. Such a relationship is provided in section 3 of this report.

All samplers, except the Gradko diffusion tube, need some kind of weather shelter for protection against rain and/or strong winds. This could be a natural shelter in the study area/city or self constructed ones. Generally the instrument producers recommend their own weather shelters. A comparison of the performance of the samplers, also addressing the effect of shelters, can be found in Gerboles et al. (2006).

Figure 1: Left: Passive sampler (Palmers tube) and housing from Passam (www.passam.ch). Right: Passive sampler (badge) with housing from Ogawa (www.ogawausa.com).

The samplers should be preferably located where air circulates freely, avoiding any form of recess but also avoiding corners of buildings or other higher than usual turbulence areas. They should be placed at about person height, or not more than 5 meters above ground to give an assessment for human exposure. The producers often recommend to have two samplers in each location to reduce uncertainty and the risk of data loss.

Table 4: List of a number of commercially available passive samplers.

Name/ Producer	**Type**	**Recommended exposure time**	**Approximate price for each sample**	**Uncertainty**
Radiello®	Radial tube	1 day to 1 week	75 NOK, without analysis	< 12%
IVL	Badge	2 -4 weeks	210 NOK, with analysis (price from NILU)	20 %
Ogawa	Badge	1 day-2 days typically ideal, but up to 4 weeks in low concentration areas	500 NOK(84 USD)for shelter, clip etc (Reusable, so a one time cost) 210NOK(36 USD) for one pad with analysis	10 %
CSIRO	Badge	1-4 weeks	250-500 NOK(50-100 AUD)	10 - 20%
Passam ag	Palmers tube	1-4 weeks	100 NOK (12,5 €) with analysis, 60 NKr (7 €) without	25%
Gradko International	Palmers tube	2 - 4 weeks	40 NOK (3,75£) with analysis	< 30%

2.2 Costs

The costs of using passive samplers for measuring NO_2 can be roughly divided into three, i.e. material cost, analysis costs and manual work. The material cost of all samplers is fairly low. The sampler itself, but also the mounting equipment and weather shelters, are simple and cost effective. The analysis is also simple and hence not expensive. The price for the material and analysis comes in the range of 40 - 500 NOK for one "station" for one sampling period. In Table 4 the prices are provided where the information was available from the producer. These should be reassessed before any monitoring is carried out. The costs for transport to the analysing laboratory could also vary to some degree, dependent on whether the producer requires the use of a specific laboratory for their product.

A considerable part of the cost is related to the manual work for setting up and changing the samplers. All types of passive samplers in Table 4 are easy to handle so this cost would be more or less independent of the material used. If the study is pre-designed and it is already decided where to place the samplers, it will typically take 2-3 working days to get 30 to 50 samplers in place, or about 30 minutes for each. Less time is needed to change or take them down. Note that the cost for designing the measurement study and treating the results by creating maps etc. is not considered here.

2.3 Exposure time

The typical exposure time is 1 - 4 weeks. The concentration levels expected can influence the exposure time as the instruments may become saturated when concentrations are high and exposures are long. Information concerning saturation levels can best be obtained from the individual producers directly but it is also possible to conduct a set of test sampling times to determine the required length of exposure. For most areas in Norway, in regard to ambient air, saturation is not likely to occur for NO_2 when using the recommended exposure times given by the producers. E.g. for the IVL passive samplers the maximum mean concentration, above which saturation starts to set in, is roughly 150 $\mu g/m^3$ for a 2 week integration period.

2.4 Comparison to standard methods

As previously mentioned it is always appropriate to carry out validation exercises when using passive samplers. This is best achieved by comparison with active samplers in the field. If this is not possible then duplicate measurements, many measurements at the one site, should be carried out to indicate the inter-variability of the samplers. Measurements in laboratory conditions may also be used but these are not indicative of real conditions of the samplers so this method is of limited use.

The aim of such comparisons is to validate, and if required correct, the passive sampling measurements. This is most often done through linear regression. In addition such comparisons allow an estimate of the uncertainty of the samples (section 2.5). Examples of such validation assessments are shown in Figure 2.

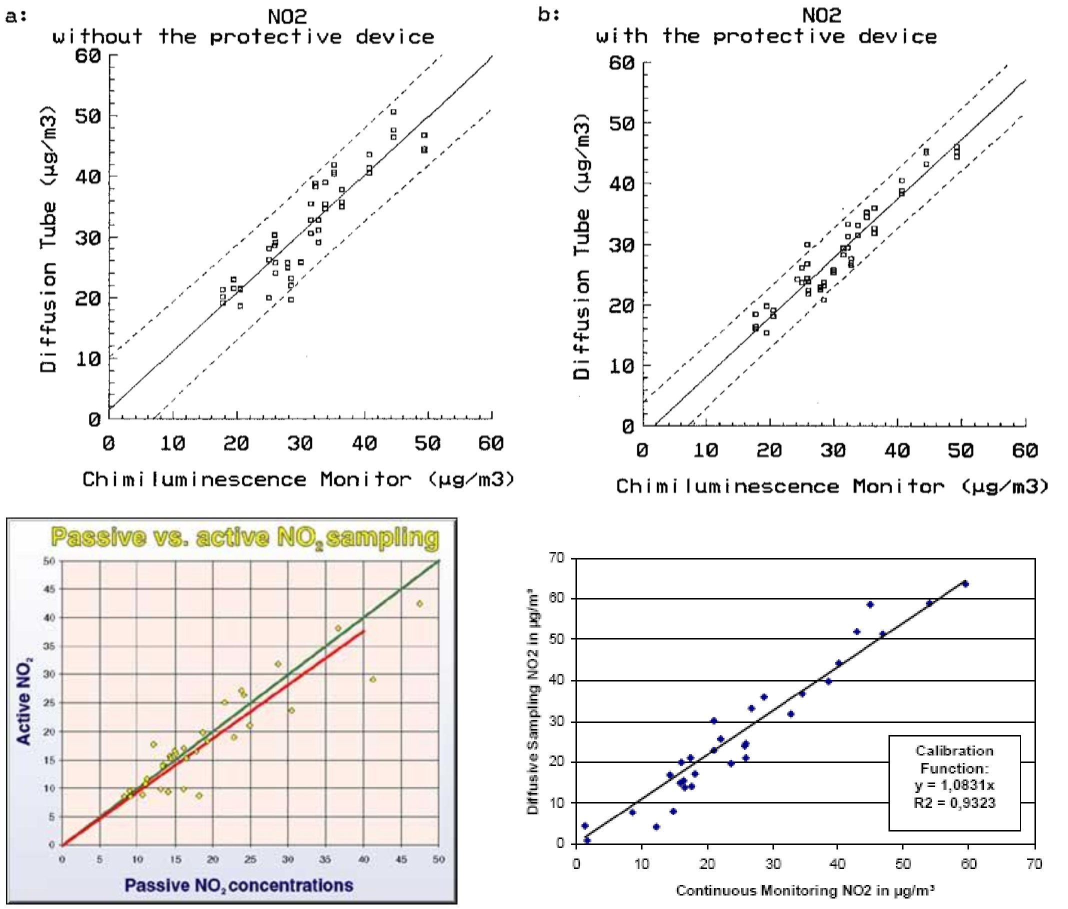


Figure 2: *Three examples of validation measurements where passive samplers are compared to active sampling techniques. The top two graphs are taken from Plaisance et al. (2002) where a study looked at the effects of weather shelters on the results. Bottom left are results from passive sampling validation exercise carried out in Oslo by NILU (www.nilu.no/AQM). Bottom right are the results from validation exercises carried out in Cyprus, taken from Pfeiffer (2006).*

2.5 Uncertainty

The uncertainty of the passive samplers is related to the actual uptake capacity and analysis procedure and may vary from instrument to instrument, from producer to producer and from time to time. It is generally recommended to carry out simultaneous measurements using standard active samplers and use these as a validation for the passive samplers (section 2.4). Several published studies have done just this. Plaisance et al. (2002) estimate the uncertainty of their passive samplers to be around 5 $\mu g/m^3$, given concentrations that vary between 15 and 45 $\mu g/m^3$. Gerboles et al (2006) carried out an extensive intercomparison study. In that paper it was found that "…the majority of diffusive samplers fulfil the 25% uncertainty requirement of the European Directive …". The producers give estimates of the uncertainty in the range of 10-30 % (see Table 4) and this can be seen as a realistic uncertainty estimate for a week long integration with a typical passive sampler.

2.6 Conclusions and recommendations concerning passive samplers

The following points can be made concerning the use of passive samplers for the measurement of NO_2 concentrations, particularly in regard to their application for the European air quality directive.

1. Passive samplers do not generally have the required level of accuracy to be used as 'fixed measurements' (< 15% uncertainty), in regard to the EU air quality directive quality objectives. As such other monitoring methods, standard continuous active samplers, will additionally be required if concentrations are found to be above the upper threshold level (> 32 $\mu g/m^3$ for annual means).
2. If passive samplers are to be used for the determination of annual means in areas where NO_2 concentrations are below the upper threshold level, but above the lower threshold level, then an assessment of their uncertainty will be required to determine if the measurements fulfil the directive quality objectives for 'indicative measurements', i.e. uncertainty < 25%. This should be done by comparison with standard 'fixed measurements'.
3. The low cost level of passive samplers and their reasonably low uncertainty make them suitable for carrying out mapping activities. However, if such maps are to be used for reporting air quality to the European commission then the above points, concerning level of uncertainty, need to be taken into account.

3 Conversion of passive sampler measurements to annual mean and percentiles

In section 2 passive sampling for NO_2 was described and the quality of the measurements assessed. Under most practical applications it will not be necessary or even possible to conduct continuous measurements to determine annual mean concentrations of NO_2. It is also not possible to directly measure percentile concentrations using passive sampling. This section deals with the conversion of a limited temporal sampling coverage to annual mean and percentile concentrations.

3.1 Conversion to annual mean

To determine the annual mean concentrations of NO_2 according to the EU quality objectives for 'fixed measurements' a temporal coverage > 90% is required (Table 3). In situations where the concentrations are found to be below the upper threshold then 'indicative measurements' may also be used. These require a temporal coverage > 14 %, meaning one day per week or 8 weeks a year as a minimum. Passive sampling is most often carried out for 'representative' periods, a few weeks of the year, and these representative periods are converted to annual means either by comparison with other fixed continuous measurements, when these are available, or by an assumption that the limited sampling can be used to replace the annual mean. This last is most susceptible to error as it is not known how representative sampling periods are if they are not directly comparable with continuous measurements. This is particularly the case when NO_2 levels are episodic. In Norway many high concentration days are during winter time so if measurements are only carried out during the winter an overestimation is likely.

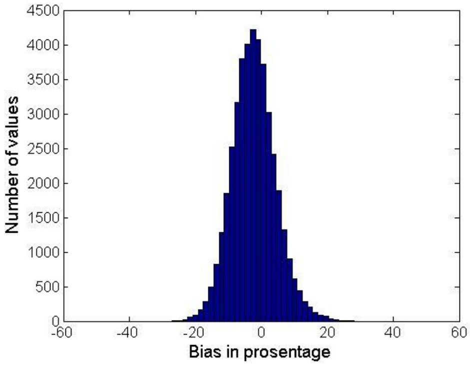
To assess the uncertainty in using a number of sampling periods that do not cover an entire year, a study has been made to evaluate the effect of sampling frequency and period on the estimate of annual mean concentrations. Three different sampling strategies have been considered:

1) One week continuous sampling, once a month for the whole year (12 weeks total coverage)
2) Two weeks continuous sampling in spring, summer, autumn and winter (8 weeks total coverage)
3) Four weeks continuous sampling in winter (December - February) and four weeks continuous sampling in summer (June - August), (8 weeks total coverage).

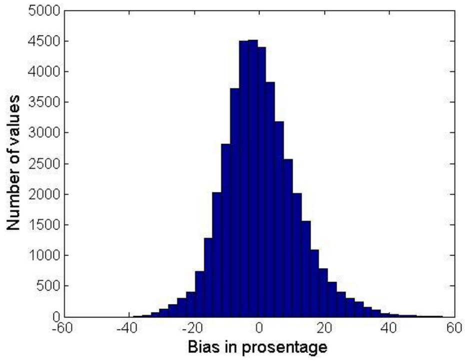
Hourly mean concentrations taken from 6 continuous measuring sites in 4 Norwegian cities have been used to test the sensitivity of the annual mean when determined using different sampling periods and frequencies, as outlined above. Only data series with more than 75% coverage of the year and with some data in all months were chosen for the study. The annual mean concentration for each station based on all hourly values (X) has been compared with an ensemble of randomly selected yearly averages (Y_i), where '*i'* is one of the realisations of the ensemble, determined using a random selection of sampling periods as outlined in the sampling strategies 1 – 3 above. The annual mean bias (M_i) between the actual mean concentration X and one of the calculated ensembles, Y_i, is then calculated as:

$$M_i = (Y_i - X)/X$$

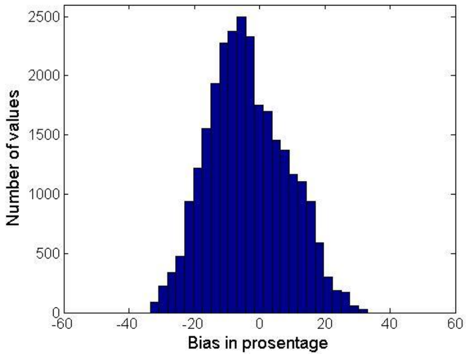
All the biases for all stations were averaged and the resulting distribution of biases were determined for roughly 30 000 realisations of the ensemble. The frequency distribution of the ensembles, for the three strategies outlined above, is shown in Figure 3. These figures show that there can be significant error in the annual mean concentration when limited sampling is undertaken.



1) One week per month



2) Two weeks every season



3) Four weeks in winter /summer

Figure 3: Histograms of the ensembles used to assess the 3 different sampling strategies. These show the number of occurrences, from roughly 30 000 randomly sampled realisations, of biases between the different realisations and the actual annual mean concentration.

The results are summarized in Table 5 where the average and standard deviation of the ensembles for the 3 different sampling strategies are given. The averages are close to 0, which is to be expected given the large number of ensemble members used (up to 30 000). A small standard deviation indicates a small uncertainty in determining the actual annual mean concentration. The smallest standard deviation in the annual mean bias occurs when using the one week per month sampling, strategy 1, which is expected as this would best include all the possible meteorological and emission conditions. The largest deviation is for the four week summer/winter season sampling, strategy 3. One can see from the histograms that the worst case sampling approaches a 40% difference between the estimated and the real annual means.

Table 5: Average mean bias and standard deviation for the 3 different sampling scenarios.

Sampling strategy	Averaged mean bias (%)	Standard deviation (%)
1) one week per month	-2.2	6.5
2) two weeks every season	0.5	11.2
3) four weeks in winter /summer	-3.6	12.0

3.2 Conversion to percentiles

In addition to the annual mean the EU air quality directive also sets a limit value on the 19'th highest hourly mean concentration (< 200 $\mu g/m^3$). This is roughly equivalent to the 99.8 percentile and is intended to reduce the effect of short term critical exposures to NO_2. This cannot be directly measured using passive samplers. However, there is often a high correlation between annual means and percentile values for air pollutants and so it may be possible to estimate these higher percentiles from annual mean measurements, something that can be determined using passive sampling.

To estimate percentiles from annual mean concentrations, data on annual mean and percentile concentrations was retrieved from the Airbase database (Airbase, 2008). NO_2 concentrations from the 3 Nordic countries of Norway, Sweden and Denmark were used, available between the years 1982 and 2006. In total around 230 individual station statistics were obtained.

In Figure 4 the 95'th and 98'th hourly mean percentiles are plotted as a function of annual mean concentration. These plots show that there is a close correlation between these percentile bands and the annual mean concentration.

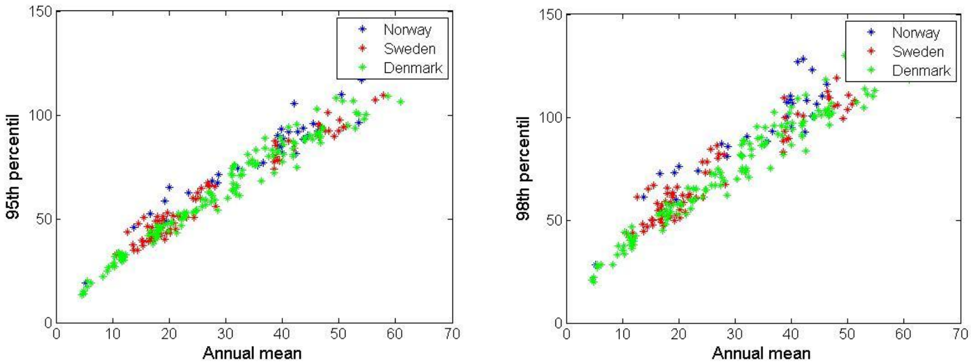


Figure 4: Relationship between the annual mean concentration of NO_2 and the 95'th (left) and 98'th (right) percentiles. In total around 230 inidvidual annual statistics are used from the Nordic countries of Norway, Sweden and Denmark.

In Figure 5 the 99.8'th percentile (19'th highest hourly mean concentration) is plotted. This shows a much larger scatter than do the lower percentile plots, particularly for the Norwegian sites, indicating that the high percentiles are the result of infrequent and unpredictable episodic occurrences. In Oslo for example measurements have shown that a single day, due to adverse meteorological conditions, can lead to a large number of hours over the 200 $\mu g/m^3$ limit.

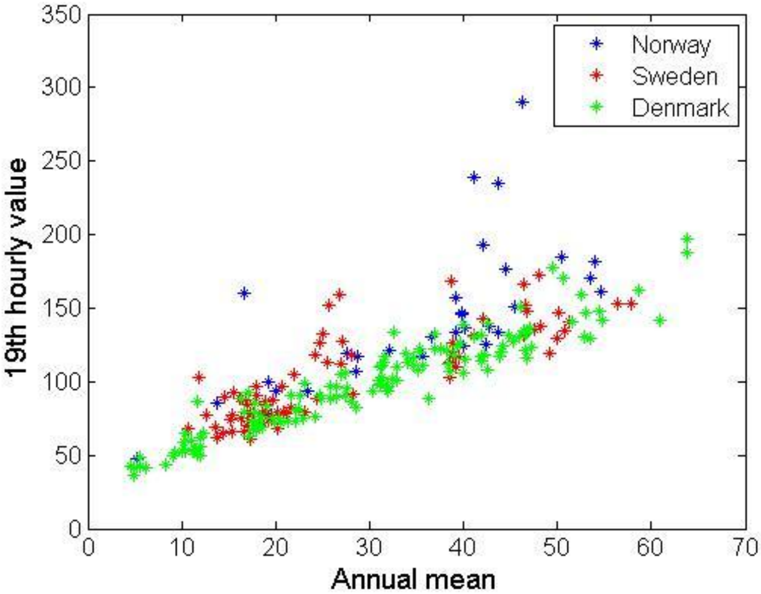


Figure 5: Relationship between the annual mean concentration of NO_2 and the 99.8'th percentile (19'th highest hourly mean concentration). In total 230 inidvidual annual statistics are used from the Nordic countries of Norway, Sweden and Denmark.

It is thus difficult to determine any clear relationship between the annual mean and 19'th highest hourly mean NO_2 concentrations at Norwegian sites. However, it is important to note that exceedances of the percentile limit only occur at stations with annual means above the limit value. I.e. nowhere is there a percentile exceedance where there is not already an annual mean exceedance.

3.3 Conclusions concerning the use and application of passive sampling measurements

There can be a range of conclusions and recommendations based on the results presented in this section, dependent on the application for which the passive samplers are to be used. If passive sampling measurements are to be used in regard to the EU air quality directives then the following conclusions and recommendations are given:

1. If passive sampling is to be carried out intermediately throughout the year then it is recommended to carry out measurements for at least one week in every month, in order to reduce the uncertainty of the limited temporal sampling.
2. Sampling for lesser periods or for more selective times, e.g. seasonally, will lead to larger uncertainty in the annual mean concentration (see table

5). If this increased level of uncertainty is considered acceptable, e.g. in regard to the costs of implementation, then 2 week sampling every season is recommended above just winter and summer sampling for 4 weeks each.

3. The assessment of hourly mean NO_2 percentiles from a number of Nordic countries indicates that the annual mean limit value for NO_2 (40 $\mu g/m^3$) is exceeded before the percentile limit value (200 $\mu g/m^3$ hourly mean less than 19 times). To indicate exceedance of both limit values it is sufficient to determine the annual mean concentrations. However, if these are above the upper threshold limit then continuous monitoring should be employed in any case.

4 Review of mapping methods

In this section a review of methods used for the mapping of air quality is provided. The focus of the review is on the use of passive samplers but other examples may be equally applicable even though passive sampling has not been used. A number of these, and other, examples can be found in the ETC report (Denby et. al., 2005) that reviews mapping methods in general.

4.1 Point wise representation

Any set of spatially distributed data may be represented in a map as points in space. This is a trivial method for representing a spatially varying concentration field but requires no added information or methods to provide the data. The representation may be made using coloured shapes, the colours used to represent concentrations on a colour scale. The size of the shape, usually circles, may also be used to indicate concentrations or alternatively the size may be used to indicate the representative area of the measurement.

In Figure 6 an example is given of point wise representation. It shows the monthly mean concentrations of 50 NO_2 passive samples taken in Dakar (Sivertsen et al., 2006). Within the city area there is large variations in concentration levels, dependent mostly on the proximity of the samplers to emissions sources, primarily traffic.

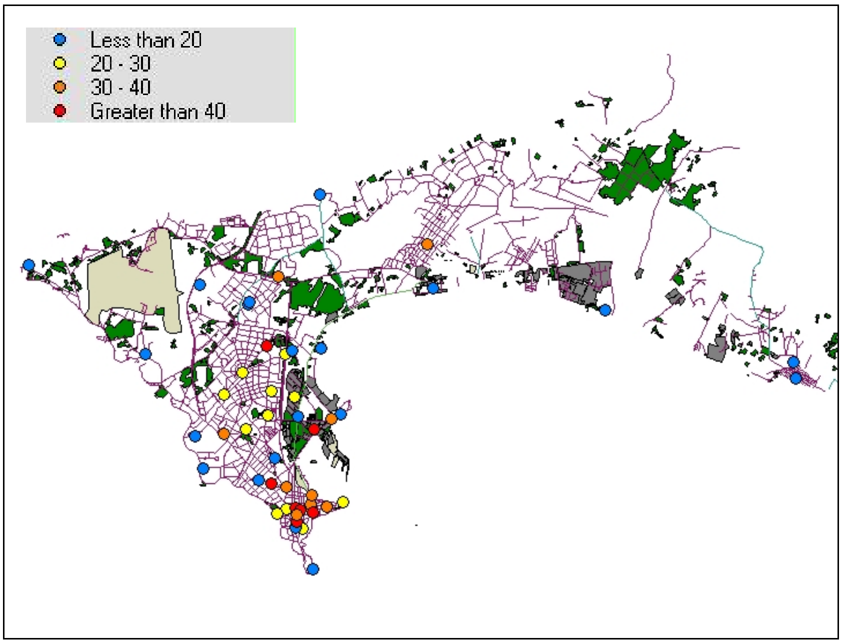


Figure 6: *Geographical distribution of the monthly mean NO2 concentrations measured by passive samplers in October 2005 for Dakar. Taken from Sivertsen et al. (2006).*

In Cyprus a large campaign was carried out of passive sampling for NO_2. 270 passive sampling sites were used. Various interpolation methods were applied but maps of NO_2 concentrations at points were also provided (Figure 7).

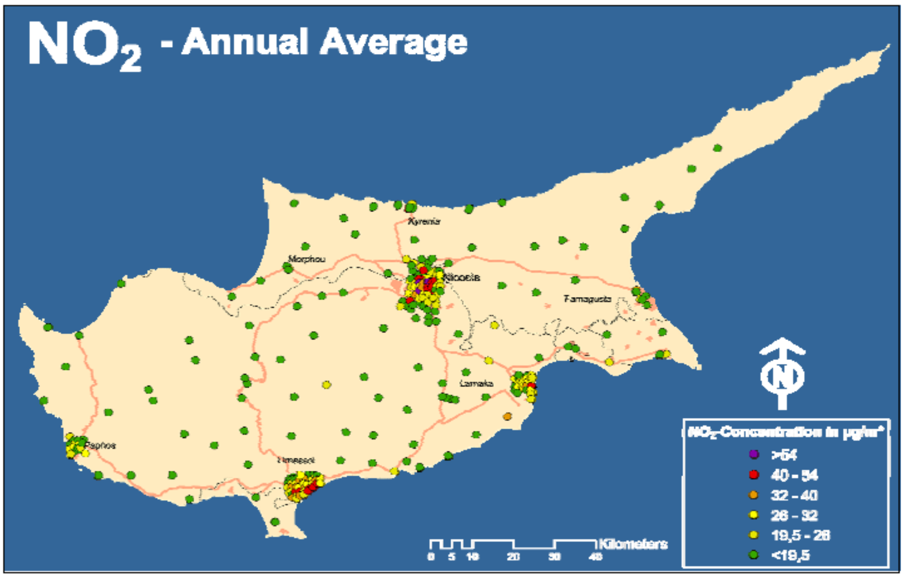


Figure 7: *Mean annual diffusive sampling results at 270 sampling sites in Cyprus (summer 2002 to summer 2003). Taken from Pfeiffer (2006).*

4.2 Subjective methods

Given any number of sampling measurements it is possible to use subjective methods to obtain a contour map of the concentrations. This involves expert understanding of the meteorological and emission fields in the area involved. It basically involves the drawing, by hand, of concentration contours which would seem reasonable given the level of expert knowledge available. It is difficult to assess the uncertainty of such mapping methods. Two examples of such maps are provided for Oslo (Figure 8) and for Bergen (Figure 9).

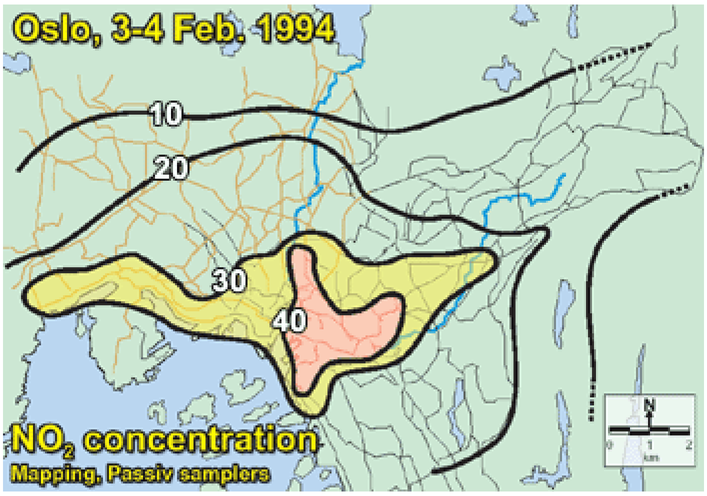


Figure 8: *24 h average NO_2 concentration distribution measured in Oslo on 3 - 4 February 1994 with 20 passive samplers. Contour lines are drawn by hand (www.nilu.no/AQM).*

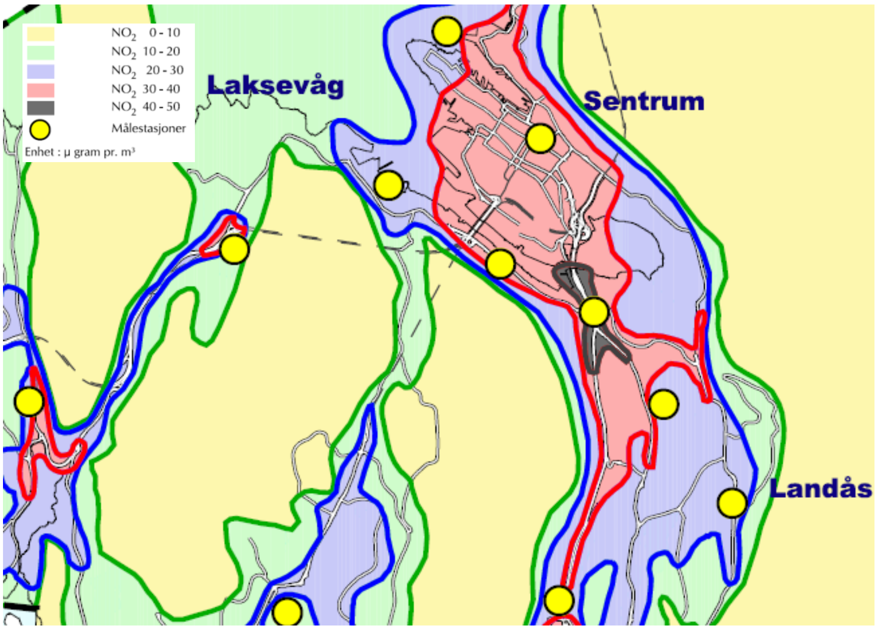


Figure 9: *Section of the NO2 map for Bergen (2005/2006 winter mean). The map is based on earlier maps (1998/1999) using 27 passive samplers and the 2005/2006 map used an additional 18 passive samplers. Exposure times for the passive samplers was 1 month and these were converted to winter means by comparison with standard continuous measurements. Taken from the Bergen annual air quality report (Bergen, 2008).*

4.3 Objective interpolation methods using air quality data alone

There are a range of methods available for interpolating spatially distributed data that can be applied for mapping air quality using passive sampling. These fall into two general categories.

1. Geometrical methods
    a. Inverse distance weighting
    b. Bilinear interpolation
    c. Radial basis functions
    d. Cubic splines
2. Geostatistical methods
    a. Kriging methods
    b. Optimal interpolation

Geometrical methods use a variety of mathematical functions to aid the interpolation. The interpolation is not based on any prior knowledge or understanding of the system. These methods are often used for visualisation of spatial data, providing smooth plots. They can be found in standard GIS software.

Geostatistical methods make use of spatial statistics, derived from the available data, to make a statistically based interpolation in space. I.e. given all the available data and it's spatial distribution then these methods provide the 'most likely' value at an interpolated point in space. Ordinary kriging is the most often used method for such interpolations and this, along with other variations of kriging, are also available on standard GIS software. One non-trivial advantage of using geostatistical methods is that they also provide an estimate of the uncertainty of the interpolation, given the available data. This information is important for understanding the uncertainty of any map.

A number of examples are available in the literature where passive sampling has been used and interpolated in space using these objective methods. The following examples show some of these.

As part of the EU life project MACBETH (MACBETH, 1999) passive samplers were distributed around a number of European cities to determine Benzene levels for exposure calculations. Padua was one of these cities. A planned campaign was intended to give higher resolution coverage in the city centres (Figure 10). Interpolation of the data was carried out using one of the geometrical methods, though this was not specified.

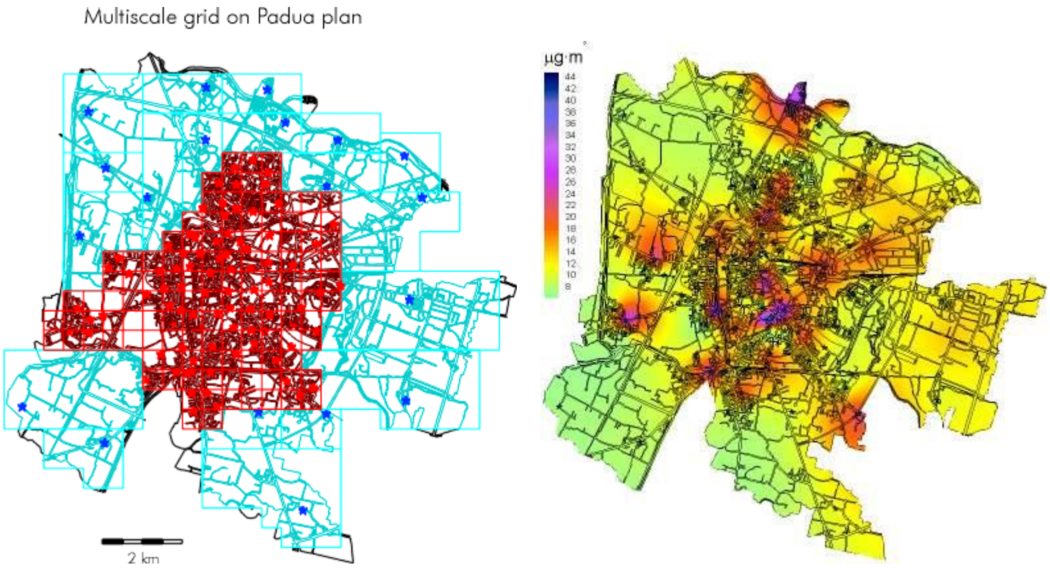


Figure 10: Multi-scale grid drawn onto Padua city map (left). The mesh size varies from 500 m in the town centre (red area) to 2000 m in the suburbs. The stars locate the sampling points which were placed in each of the grids where possible. On the right is the interpolated concentration of Benzene. Geometrical methods were used for the spatial interpolation.

The township of Haifa, Israel has a number of air quality stations available that measure continuously throughout the year. A study was carried out (Yuval and Broday, 2006) to assess the spatial patterns of a number of pollutants in the Haifa Bay region. For NO_2 10 stations were available for the annual mean mapping. Kriging was used for the interpolation (Figure 11).

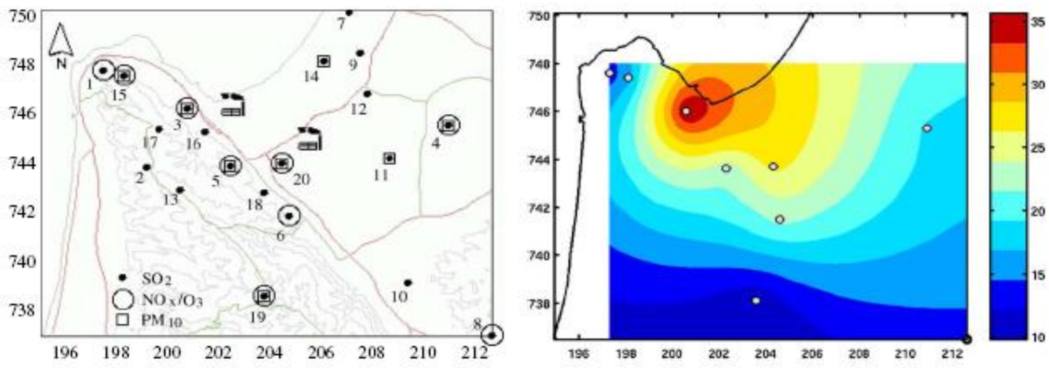


Figure 11: Showing (left) the position of monitoring stations, major industries and major roads in the Haifa Bay area and (right) annual mean interpolated NO2 concentrations in µg/m3. Taken from Yuval and Broday (2006).

A passive sampling campaign, using 30 sites, was carried out in Pancevo, Serbia, in 2006. (Alegrini et al., 2007). Within this campaign a range of pollutants were measured and their concentrations interpolated in space using inverse distance weighting as an interpolation technique (Figure 12).

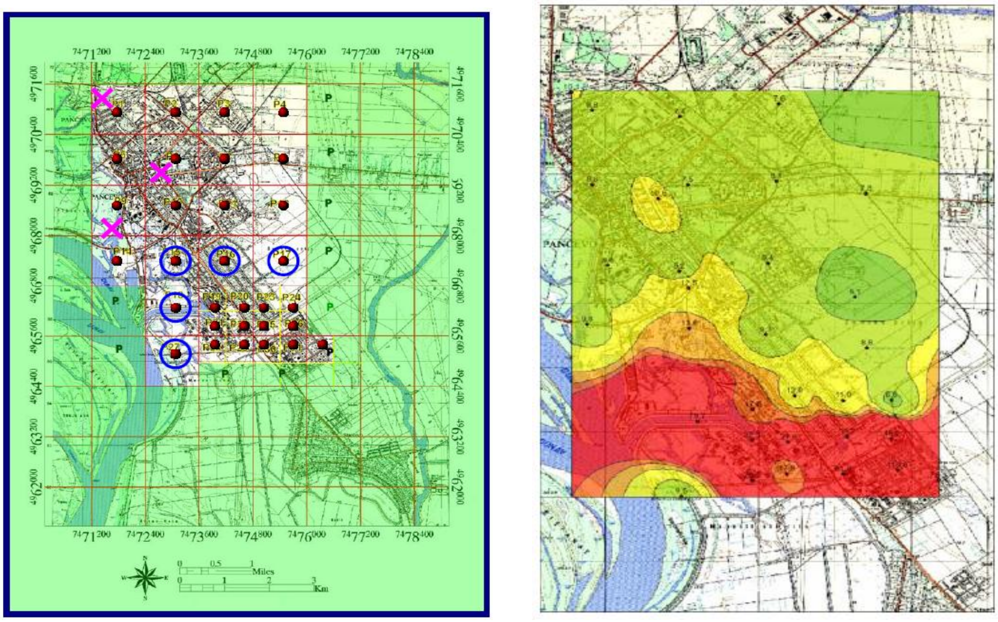


Figure 12: (Left) Passive sampling monitoring network for the Pancevo campaign (the red dots represent the sampling points, the pink crosses represent the hot spots and the blue circles identify the locations at which duplicates were exposed). (Right) Annual mean Benzene concentrations interpolated using inverse distance weighting. Taken from Alegrini et al. (2007).

Plaisance et al. (2002) carried out a 2 week passive sampling campaign in the French agglomeration consisting of Lille, Roubaix and Tourcoing. In all 145 measurements of NO_2 were made, covering an area of 300 km^2. The sampling network for these applications was set up so that there was one site per 1 km^2 in the urban centres and one site per four km^2 in the suburban and rural areas. The measurements were all situated at least 50 m from significant sources of air pollution, such as busy traffic roads. Kriging was used to interpolate the data. The aim of the study was to determine the long term background levels for human exposure estimates (Figure 13).

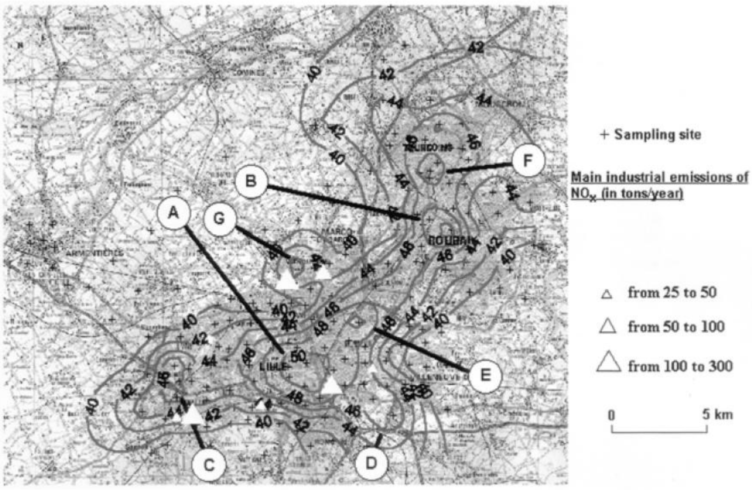


Figure 13: Showing the NO_2 contours taken from a 2 week sampling period in the French agglomeration consisting of Lille, Roubaix and Tourcoing. 145 samples were taken and the interpolation was made using kriging. Taken from Plaisance et al. (2002).

In Cyprus a large number of passive samplers were distributed in a number of major cities and over the entire island. Different interpolation methods were applied including the use of inverse distance weighting. The resulting maps of annual mean NO_2 concentrations are shown here for both the entire island and also the city of Limassol (Figure 14). For all of Cyprus 270 stations were used, see figure 7. For Limassol 20 passive samples were used.

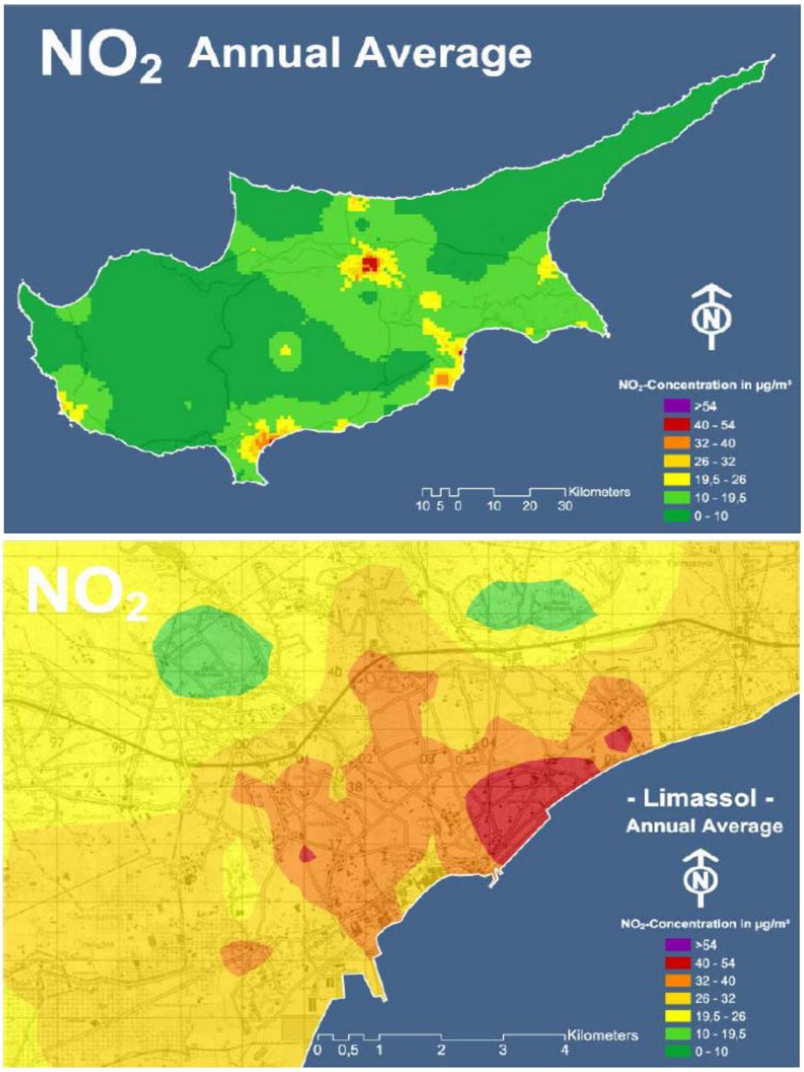


Figure 14: (Top) Inverse distance weighting of NO2 annual mean concentrations using 270 passive samplers for all of Cyprus. (Bottom) Close up of the city of Limassol using 20 passive samplers. Taken from Pfeiffer (2006).

4.4 Mapping methods using supplementary data

There is often some degree of correlation between air pollutant concentrations and a range of other physical or meteorological parameters. Most important is the proximity to, and rates of, emissions. Clearly close to roads, the major source of pollutants such as NO_2, the concentrations of pollutants will be highest. In addition to emissions, meteorology plays a strong role in both the general and in the local meteorological environments. E.g. concentrations will be higher in street canyons than in open road systems, given the same traffic volume, due to the local wind circulation. Channelling of winds through valleys and capping inversion will play a significant role in the dispersion of the pollutants. Most of these factors are combined in air quality modelling but these parameters, or representations of these parameters, may also be used in a more statistical manner to improve the air quality mapping. Importantly the spatial distribution of these parameters should be well known so that they can be used to help interpolate the air quality data, allowing higher resolution mapping to be achieved.

A number of varied studies have been carried out to this affect and they include methods such as:

1. Co-kriging: where the correlation between some other relevant parameter is used in the spatial statistical description, e.g. altitude or another pollutant.
2. Regression: Statistical relations are built up between known spatially distributed data, e.g. proximity to roads, traffic volume, wind speeds, etc. Multiple linear regression is often used to determine these relationships.
3. Neural networks: Similar to regression, a number of supplementary parameters are used to ‘train’ a neural network to help produce maps based on these data.

The following examples apply some of these techniques.

Mesquita et al. (2007) created maps of NO_2 at 100 m resolution by combining 100 passive sampler measurements with 9 automatic monitors, traffic emissions, and industrial emissions. One month sampling was used in both summer and winter. The passive sampling data were combined with the fixed monitoring data, through regression analysis, to determine annual mean of the passive samplers. These were then combined with the emission data using multiple linear regression. To combine these spatial data the concept of influence zones was applied, so that only emissions within 500 m of a passive sampler were used in the regression. In addition to the linear regression method ordinary kriging was also applied. The results are shown in Figure 15.

Comparison of these two methods indicates the typical results obtained when using interpolation only and when using spatially distributed supplementary data to represent air quality levels. There is clearly more spatial detail available when using the supplementary data and in many ways it appears more physically representative of reality, though some features such as rings around measurement points are less representative. However, without a comprehensive assessment of the uncertainty it is not possible to come to firm conclusions concerning the validity of such maps.

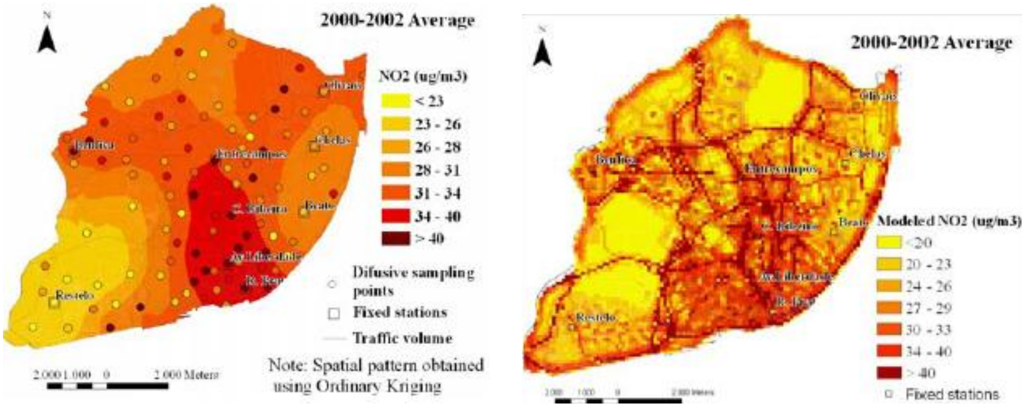


Figure 15: Results of the mapping of annual mean NO2 in Lisbon using ordinary kriging (left) and multiple linear regression of emissions (right). 100 passive sampler sites were used. Taken from Mesquita et al. (2007).

Jerrett et al. (2003) also carried out a similar activity in Toronto, Canada, where they created maps of NO_2 by using multiple linear regression of 95 passive sampler measurements with 9 automatic monitors, traffic counts, wind speed/direction, distance from roads, population density, number of dwellings and land use data. In addition to the linear regression method ordinary kriging was also applied. The results of this study are shown in Figure 16.

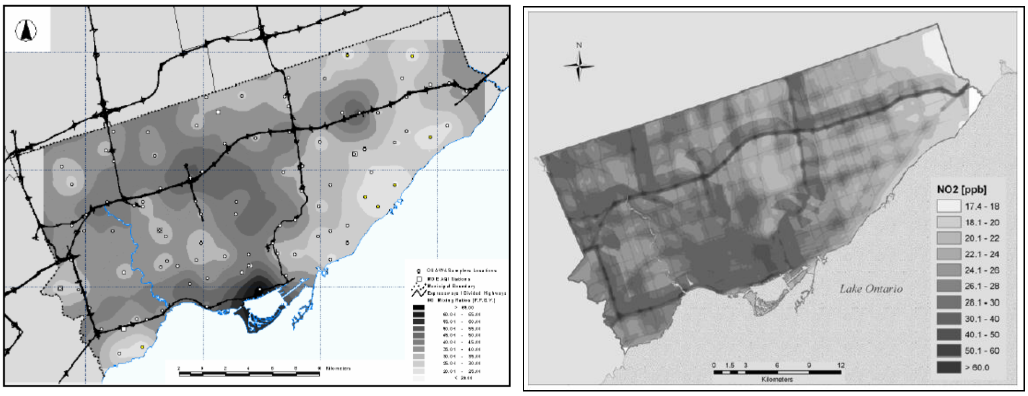


Figure 16: Resulting maps of NO2 from the study by Jerrett et al. (2003). Left is the kriged surface of NO2 using 95 data points. Right is the map created using multiple linear regression with 8 different land use parameters, including wind direction.

Another methodology that makes use of supplementary spatially distributed data are neural networks. Given a range of data such networks can be trained to provide pollutant concentrations based on this supplementary data only. Such a study was carried out in Cyprus (Pfeiffer, 2006) for NO_2 (see also Figure 7 and Figure 14) where emission data, population density, and geographical co-ordinates were used for the training. The resultant map is shown in Figure 17.

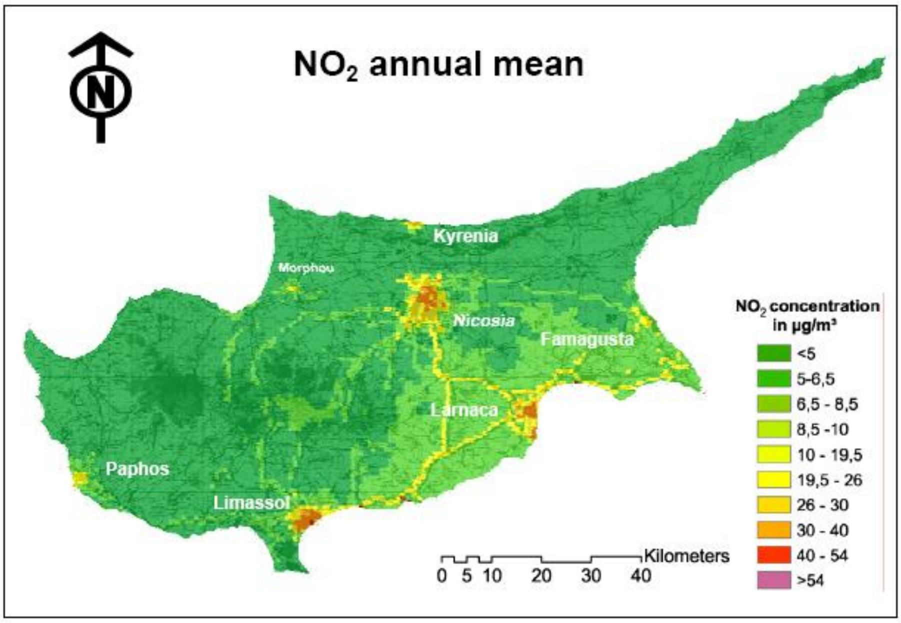


Figure 17: NO_2 distribution for Cyprus as calculated by a neural network trained with coordinates, population density and NOX emissions as input variables. 270 passive sampling sites were used for the training. Taken from Pfeiffer (2006).

Lloyd and Atkinson (2004) make a comparison of interpolation methods including inverse distance weighting, linear regression, ordinary kriging, kriging with an external drift and simple kriging with a local varying mean. Emission fields (at 1 x 1 km resolution) of NO_X are used as supplementary data in the interpolations in order to calculate NO_2 fields for the United Kingdom, using a methodology described by Stedman et al. (2001). The conclusion from that study is that simple kriging with a locally varying mean gives the best results, the local mean being determined from the regression with the NO_X emission data (Figure 18).

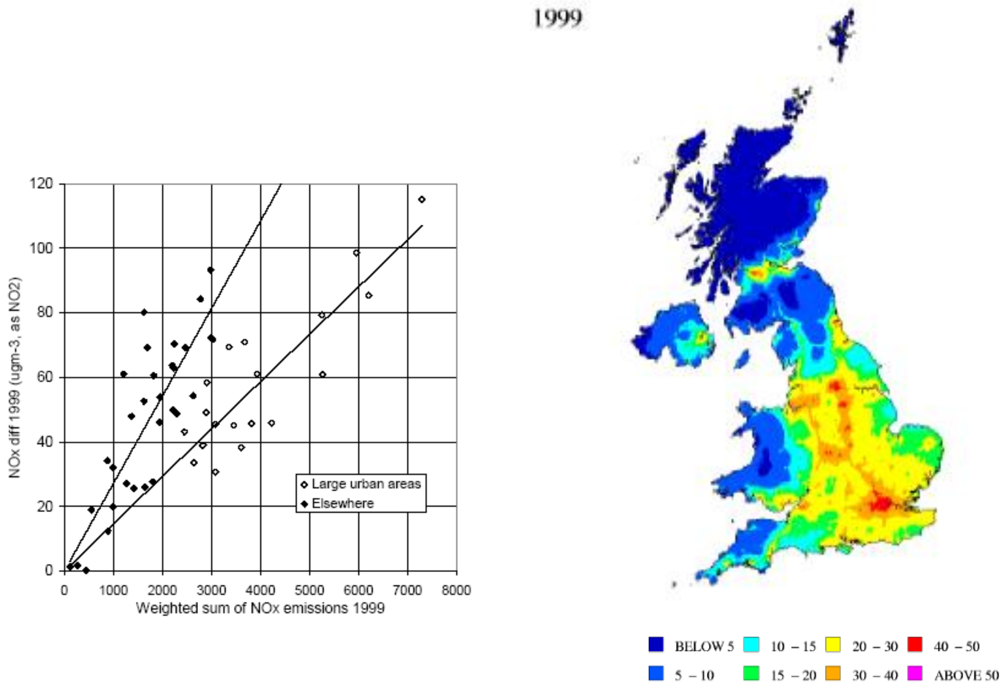


Figure 18: *(Left) The relationship between ambient NO_2 concentrations against the weighted sum of local emission (Stedman et al., 2001). These data are used to establish the regression coefficients applied to produce the air quality map (right). Measurements for urban and elsewhere (rural) have been separated and given different regression coefficients.*

The above examples combine observations with spatially distributed fields, essentially providing factors for the supplementary data so that when they are combined they match as closely as possible the observed concentrations. This type of method does not necessarily lead to a map that gives exactly the same concentrations as the measured data. In addition to the above methods some form of residual interpolation may be applied to the maps. This involves subtracting the mapped concentration from the measured concentration, giving a difference. Interpolation of this difference, using geostatistical or geometric methods, may be then added to the map so that the mapped concentrations at monitoring sites will then correspond to the observed concentrations, with some form of interpolation in between. Such residual methods were applied in the Lloyd and Atkinson (2004), study described above, and have also been applied in combination with modelling data. Three examples of this are provided in the following section (4.5).

4.5 Mapping methods using air quality models

Air quality models combine many of the relevant 'supplementary' data described in the previous section and use the physical relations between them to calculate concentration levels. In this way air quality models should provide the best basis for any mapping activity. Of course air quality models require a range of input data, particularly emissions and meteorology, before they can be effectively used. Often this information is not directly available and the methods described in the previous section (4.4) are easier to apply. If air quality models are available then they can be treated in a similar way to any other form of supplementary data and similar methods to those described in section 4.4 may be applied.

Combining observations with models is most often applied on large scale regional models. For example, maps of Europe have been constructed using the unified EMEP model by both Tarrason et al. (1998) and Horálek et al. (2007). Both methods involve the use of residual interpolation. In the first, radial basis functions are applied to the residual concentrations and in the second ordinary kriging is applied after multiple linear regression of the observations with the model and other supplementary data. The results of such mapping is shown in Figure 19 and Figure 20.

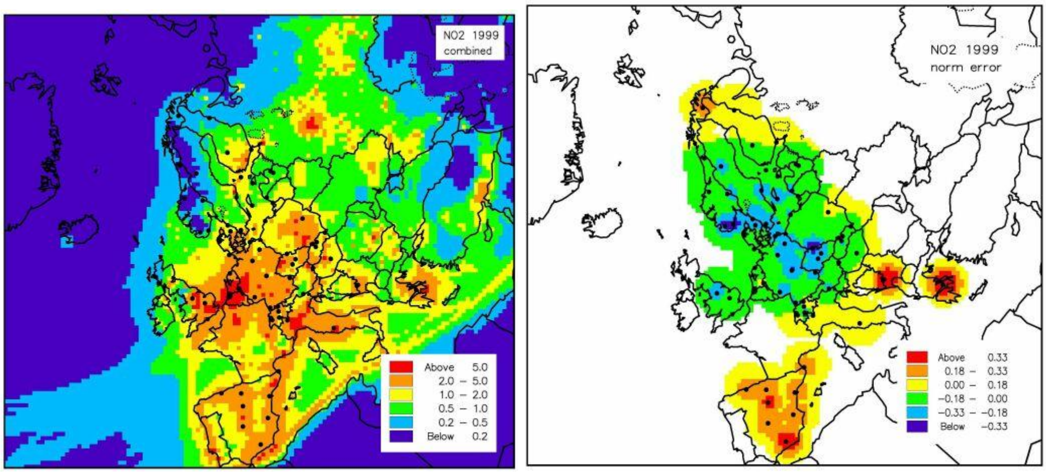


Figure 19: Example of interpolation field using the methodology from Tarrason et al. (1998) for yearly average NO_2 concentrations. Left: the final interpolated field where the interpolated difference field has been added to the model field. Right: the interpolated difference field calculated by subtracting the model field from the observations and interpolating using radial basis functions.

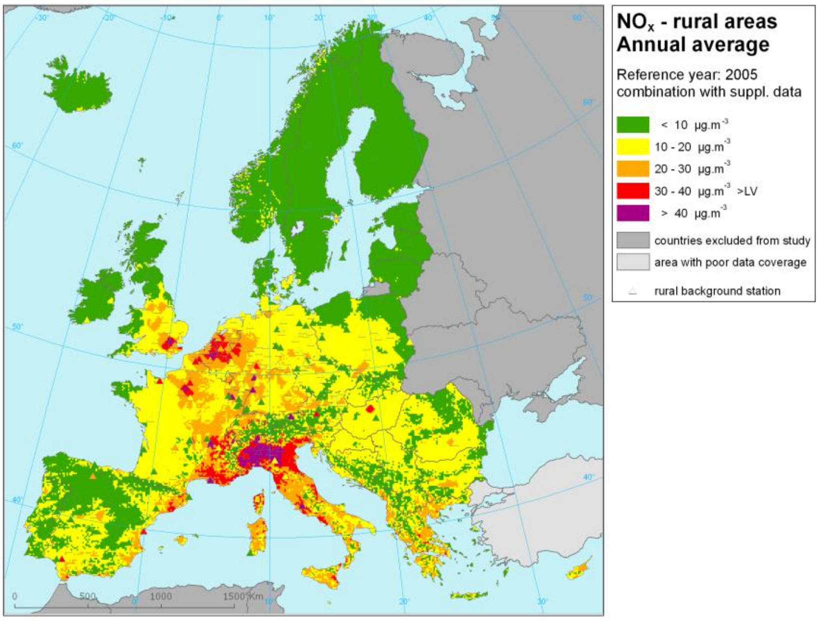
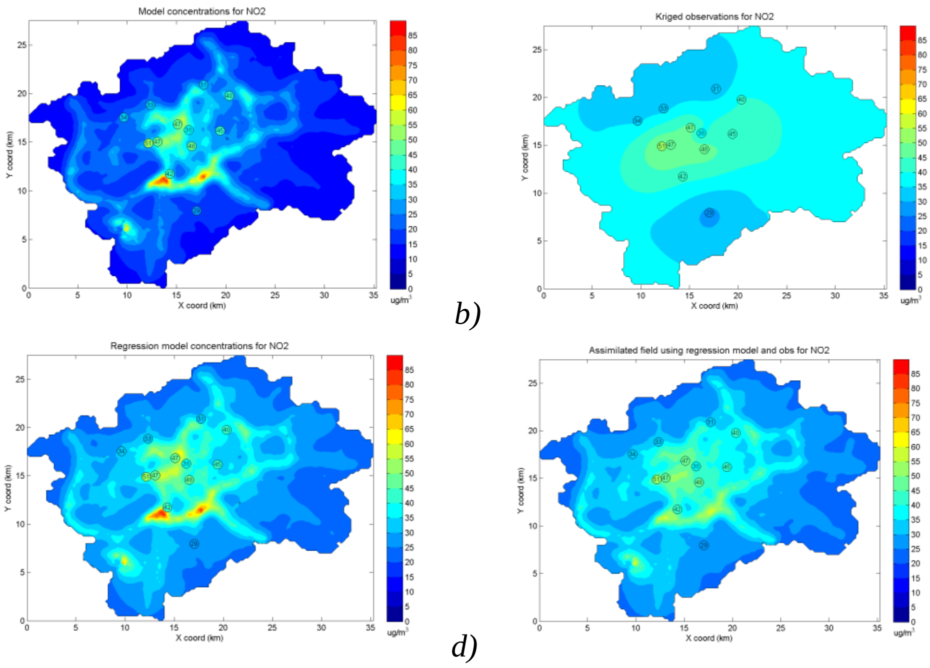


Figure 20: Results of the mapping of annual mean NO_X using rural background stations in Europe. Residual kriging has been carried out after regression of the observations with the Unified EMEP model, altitude, wind speed and temperature. Taken from Horálek et al. (2007). Also shown in the map are the observations themselves, triangles.

Some studies have used similar methods for calculating concentrations on the urban scale. In Denby and Pochmann (2007) linear regression of observations with dispersion model calculations were used to map NO_2 concentrations in Prague. Regression was used as the basic map before being combined, using Bayesian methods, with the kriged observed concentration field. The step by step combination of model and observations, using 12 sites, is shown in Figure 21. In that study maps of the uncertainty in the calculations were also produced.



a) b) c) d)

Figure 21: a) Modelled NO_2 concentrations and observations (numbered circles). b) Ordinary kriging of the observations. c) Model fields after regression with observations. d) Weighted combination of fields (b) and (c) using a Bayesian approach. Taken from Denby and Pochmann (2007).

4.6 Conclusions and recommendations concerning the review of mapping methods

From the review it is clear that a wide range of methods are available for mapping the concentrations of air pollution, when using both active and passive samplers. The following conclusions and recommendations can be made, based on the current review:

1. Mapping studies have been carried out were up to 270 passive samplers have been used. Generally the studies use < 100 samplers and this would appear to be the most feasible upper limit to the number of passive samplers that can be effectively applied.
2. Methodologies that use the interpolation of measured concentrations alone can only provide maps at a resolution similar to the spacing between monitoring sites. As such the placement of monitoring sites should match the spatial representativeness of the map resolution. Samplers placed at hotspots will, through interpolation, generally overestimate their sphere of influence.
3. Methods that include spatially distributed supplementary data have the potential for providing maps at a much higher spatial resolution than just the interpolation of the observational data alone. Though such maps provide an improved level of detail their representation of the physical reality is not guaranteed.
4. Mapping methods that use modelling provide the best basis for the mapping, when combined with monitoring data, provided the models have

the required resolution. Models provide the best physical description of the spatial distribution of pollutants.

5. Subjective methods for interpolation are difficult to assess and cannot be recommended except for visualisation purposes.
6. Few studies include an analysis of the uncertainty of the maps, though this should be provided. Cross-validation is a recommended method for providing a general assessment of the uncertainty of the maps.

5 Passive sampling networks for mapping

In this section the design of an appropriate passive sampling network is discussed. Firstly looking at sampling for screening purposes, for mapping and including the micro placement of samplers.

5.1 Sampling network for screening purposes

As described in Section 1, passive samplers have most often been used for screening studies. Such studies do not necessarily have mapping as one of their goals, rather the need to characterise the different air quality environments that are likely to be found in any given area. For screening the aim is to sample a variety of representative areas, including the following types of environments:

a) *Rural or regional background:* Sites outside and upwind of the city of interest
b) *Urban background:* A variety of sites within the city that are not directly affected by individual sources
c) *Industrial sites:* Downwind of major industrial sites at a variety of distances
d) *Major highways:* At various distances along major roads
e) *Roads:* A variety of major and minor roads at curb side
f) *Street canyons:* On either side of street canyons
g) *Others:* Near shipping or airports etc. where emissions are expected

When carrying out such studies it is always important to have an understanding of the meteorological situation, with particular regard to the dominant wind directions. Placing samplers at sites downwind of the dominant wind direction will ensure exposure of the samplers to the emissions. More information concerning the planning and network design of passive sampler screening studies can be found in a range of NILU reports, e.g. Sivertsen (2003) and Sivertsen et al. (2006).

5.2 Sampling network for mapping purposes

As can be seen in section 4, monitoring networks for mapping are designed differently to those for screening purposes. The major features of the monitoring networks for mapping are:

a) Samplers are placed systematically, in a grid like fashion
b) The density of samplers increases where the expected concentration gradients are highest, i.e. in areas with large numbers of emission sources

c) Automatic stations are often used to establish relations between limited integration times for passive samplers and the long term means.

Before designing a sampling network for mapping it is important to understand the goal of the map and to know the mapping method that will eventually be used. In cases when only samplers are to be used then a well designed grid is required, where the spacing between samplers is similar to the area they are intended to represent. This can be large for rural or urban background levels (1 – 10 km) but may be very small near traffic or industrial sites (10 – 100 m).

When supplementary data sources are used together with alternative interpolation methods, such as regression, to map the concentration fields (section 4.4) then the network can be designed differently. Monitoring sites are in this case chosen to be representative not just of positions in space but also of a range of environments, e.g. close to major or minor roads, on top of hills, in valleys, etc. In this way the supplementary data can be more directly associated with the monitoring data. E.g. if topography is considered to be important then it would be important to sample at a range of altitudes, to provide information for the regression. Similarly if distances from roads are a supplementary data source then sampling should be carried out at a variety of roads at varying distances, not just at curb side. In this way a regression model or neural network will have the appropriate information for correlating the supplementary data with the monitoring data.

The same is true for modelling. When modelling fields are available it is important to try to sample as wide a range of concentrations as possible, not just hotspots but also background levels. In this way any adaptation of the model field to fit the observations can be carried out over a suitable range of concentrations.

5.3 Micro-placement of samplers

Passive samplers are small and compact and do not require any power to run so they can be easily situated in a variety of places. Passive samplers do however need to be placed so that they are not directly subject to both rain and strong winds. Most manufacturers provide some form of weather cover that can be placed over the sampler. This type of cover is quite open since the sampler must be well ventilated. The alternative to the use of housings is to place the samplers under protected features, e.g. under balconies. Many people placing passive samplers use electrical or telephone poles for the sites. These are widespread and readily available, but generally only near roads.

Generally the sampling is intended to provide information for human health and so samplers should be placed at heights that are representative of human exposure. Near roads the variation of concentrations with height can be significant. Further from roads, that reflect urban background levels, samplers may be placed higher up buildings as the vertical variation of concentration is less pronounced. It is generally recommended though to try to place the samplers at the same level to avoid any vertical gradients. Practical considerations would generally require samplers to be placed at heights that are ‘out of reach’. Open park areas are often the best for sampling urban background levels.

5.4 Conclusions concerning network design

The design of any passive sampling network is dependent on its purpose and the methodology employed for the mapping. The following points can be made:

1. A network designed for screening purposes alone does not require regular spacing of samplers but requires placement in a range of representative environments.
2. A network designed for mapping, using monitoring only, requires a regular distribution of samplers. The distance between samples should be similar to the area they are representative of.
3. A network designed for mapping, using supplementary spatial data, requires the placement of samplers such that they represent a sufficient range of the supplementary data sources.
4. A network designed for mapping, using modelling data, has similar requirements to (3) above with the addition that a good range of concentrations are also required.

6 Methodological description of mapping methods

In this section recommendations are given on the application of the various methodologies available for mapping. The choice of method is dependent on a number of factors, including:

- The overall aim of the mapping exercise
- The availability of the appropriate supplementary data
- The competence for applying the interpolation methods
- The competence for applying alternative methods and models
- The available funding

Keeping these points in mind the following list indicates the most likely scenarios available for carrying out the mapping exercise.

6.1 Screening analysis

Before any mapping exercise is carried out there should be some preliminary investigation, either through modelling or monitoring screening studies. Carrying out such a preliminary study, as described in section 5.1, will identify both the need for further assessment as well as the types of environments that will require the most focus in a mapping study.

6.2 Mapping based on passive samplers alone

When supplementary data, or the competence to use these data, is not available and/or appropriate then mapping using passive samplers alone may be undertaken. If such maps are to be made appropriately and completely then it is important that the spacing between the sampling sites is similar to the representativeness of the sampling sites themselves. E.g. If the samples are intended to represent the urban background concentrations then samples need not be taken on any more than a 2 x 2 km grid. However, if mapping is to be carried out near major emission sources then concentration gradients will be higher and a much denser network will be required. For the case of major roads monitoring will be required on scales of less than 50 m to capture the gradient from the road. Mapping based solely on passive sampling is difficult for environments such as street canyons, which are box like environments, with a discrete change in concentration levels when leaving the canyon area.

Of all interpolation methods available kriging has generally been shown to yield the best results for a range of air quality applications, e.g. Johnston et al. (2001) and Horálek et al. (2007). Standard GIS software packages, such as those from ESRI, can be used to carry out kriging and a range of other interpolation methods. Kriging requires a number of parameters to be set. These are generally derived from the data itself and can be automatically carried out by most software. Kriging is based on a number of assumptions and these are not always met for the application of urban air quality. Even so, the method is robust and effective.

It is important to realise that the statistical interpolation methods, such as kriging, base the interpolation on the data provided, i.e. the spatial statistical character of the concentration field is represented by the available data. If all the data represents urban background levels then kriging will not be able to interpolate to local hotspot levels. If all the data is from traffic sites then kriging cannot interpolate to the urban background. A complete representation would be required and this is generally not feasible with monitoring data alone. This means that, for most purposes, urban background levels may be mapped using this method. Only when the density of sites is sufficiently high, surrounding a particular hotspot area, should this method be used for mapping of hotspot areas.

6.3 Passive samplers and other supplementary data

If detailed maps are to be made that represent NO_2 concentrations in complex urban environments then passive sampling alone will not be enough to cover all regions with the necessary detail. To obtain that level of detail, with resolutions of 100 m or less in areas close to major sources, supplementary spatial data is required. This will preferably be in the form of model fields but alternatively as parameters that can be correlated with the concentrations. Multiple linear regression is a straight forward method for obtaining weighting factors that can be used to build up a regression model. The most important factors are emissions and meteorology. However, there can also be a range of proxy data that can be used instead of these. These include:

- Geographical position and traffic volumes of roads
- Distances from roads
- Population distribution
- Geographical position
- Altitude
- Land use types

If a successful regression model can be built up with these, or other parameters, then this can be used to generate concentrations maps. There remains the question of uncertainty of the maps and their physical realism. For this reason dispersion models are preferred to statistical methods.

6.4 Passive samplers and modelling data

As previously mentioned dispersion modelling can also be used as a form of supplementary data for mapping. It has the added advantage over statistical methods that it describes as well as possible the physical processes that lead to dispersion, removing the need to include parameters such as distances from roads, wind speeds and directions, etc. from the list of supplementary data sources. However, models may not represent all the processes correctly and the inclusion of parameters such as altitude may be useful for the interpolation.

Another advantage of models is that even if emissions are not well known, but the location of them are, then coupling the passive sampling to the dispersion model will enable emission strengths to be adjusted to best fit the observed concentrations. Such methods have been applied in other studies, e.g. Laupsa et al. (2008), and can also be applied to passive sampling methods.

If models are to be used in the interpolation then a methodology similar to that applied by Horálek et al. (2008) or Denby and Pochmann (2007) can be applied. This methodology involves the application of multiple linear regression to the dispersion model, or to any other relevant spatially distributed data. After this residual kriging is applied. Though other methods for combining measurements and models have also been described in the literature, this particular combination has been shown to be effective and relatively uncomplicated in its implementation.

6.5 Assessing the uncertainty of the maps

Having produced a map, based on any of the methods described above, there will be some level of uncertainty in them. Uncertainty can come from a range of factors including:

- Monitoring uncertainty including the analysis, variability and conversion to annual mean concentrations
- Uncertainty due to the spatial representativeness of the monitoring data
- Uncertainty related to the interpolation method, be it kriging or based on supplementary data
- Uncertainty in the model

Due to this range of uncertainty it is recommended to base an uncertainty analysis on cross validation methods. This involves carrying out the mapping methodology repeatedly, but each time removing one sampling site at a time. The ‘predicted’ concentration at the missing site can be compared with the actual measurement. This is done for all measurement sites. Statistical assessment of this set of ‘predictions’ can then be applied and will include most of the uncertainties listed above. Typical statistical analysis of the cross-validation data set uses parameters such as root mean square error, correlation coefficient and bias estimates. Such assessments are often carried out when mapping is applied, e.g. Horálek et al. (2007); Denby and Pochmann (2007), and give an overall indication of the quality of the maps. They do not however take into account uncertainty of the measurements and this must be included as a separate source of uncertainty

6.6 Summary of the methodological recommendations

Table 6 provides a summary of the recommendations made in this section.

Table 6: Summary of the methodological methods presented.

Mapping method	Application	Network	Expertise level	Cost level
Screening	First estimate of air quality levels	A number of representative samples	General understanding of air quality and sources	low
Objective interpolation	Mapping of urban background or local hotspot	Very large number homogenously distributed for complete coverage	+ GIS expertise	Medium/ high
Interpolation with supplementary data	Complete mapping of urban area	Large number, focus on a range of sources	+ spatial statistical expertise	medium/ high
Interpolation with dispersion models	Complete mapping of urban area	Large number, focus on a range of source and concentration distributions	+ dispersion modelling	high

7 Recommendations and conclusions

Within this document a number of aspects concerning the application of passive samplers for air quality mapping of NO_2 have been discussed. These include the technical aspects of passive sampling, their use in relation to the European air quality directive, their use for determining annual means and percentiles, current practices for mapping air quality and finally recommended methods for mapping air quality. At the end of each of the sections conclusions are given. In that regard the discussion and conclusions are only considered to be valid for NO_2 and not for other pollutants that may have different characteristics and source contributions. For completeness these conclusions are summarised here.

- *Application:* Passive samplers are cost effective monitors that can be used for both screening studies and spatial mapping.
- *Uncertainty:* In regard to the European air quality directive passive sampling falls under the quality objective applicable for ‘indicative sampling’ but will generally require in field comparisons with standard techniques to guarantee this.
- *Sampling frequency:* It is recommended to sample regularly for a more certain estimate of annual mean concentrations. A minimum of one week every month for a year is a recommended duration and frequency.
- *Limit values:* Directive limit values related to percentiles for NO_2, which are based on hourly data, cannot be determined using passive samplers. In Nordic countries the annual mean EU limit value is seen to be exceeded before the percentile limit value, which means that passive sampling can be used to indicate exceedances. However, in cases where the annual mean limit value is above the upper threshold limit value then

the directive requires fixed measurements to be carried out of a predefined quality objective. This quality objective (uncertainty < 15%) is generally not met by passive sampling and so standard hourly sampling methods will be required.

- *Current practices in mapping:* There are a range of methods available for air quality mapping using passive samplers. A number of studies have applied objective interpolation methods whilst a large number have also applied statistical interpolation using both dispersion models and other supplementary data.
- *Recommended practises in mapping.* It is recommended to use dispersion models and other supplementary data for mapping using passive samplers as these provide the most complete and detailed analysis of air quality. Objective interpolation methods require large numbers of passive samplers to sample the required space effectively and the use of supplementary data without models neglects the physical processes.

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ABSTRACT
To assist in producing spatial maps of NO_2 concentrations this report details methodologies for creating such maps in a cost effective way, with the use of passive samplers. It provides an overview of passive sampling methods in regard to their costs and quality. It then provides a statistical assessment of the uncertainty in converting limited passive sampler measurements to annual mean and percentile concentrations, based on an analysis of measurement data available in Nordic countries. This is discussed in relation to the European air quality directives. It then reviews the current state-of-the-art in air quality mapping using both passive samplers and other data sources. Finally it describes and recommends different mapping methodologies.

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