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# Erarbeitung von Handlungskonzepten für eine gesunde Umwelt in städtischen Regionen unter Berücksichtigung des demographischen Wandels

von

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**Kurzbeschreibung**

Das Vorhaben wurde im Rahmen des internationalen Forschungsprojekts ACCEPTED (Assessment of changing conditions, environmental policies, time-activities, exposure and disease) durchgeführt, an dem sich insgesamt 11 Partner aus Belgien, Deutschland, Frankreich und Schweden beteiligten. Das Projekt wurde von fünf Organisationen (ADEME und ANSES in Frankreich, BELSPO in Belgien, Swedish EPA in Schweden und UBA in Deutschland) im Rahmen des ERA-ENVHEALTH Netzwerks finanziert.

ACCEPTED ist ein internationales Forschungsprojekt mit dem Ziel, zukünftige Expositionsszenarien in Städten und ihren Einfluss auf die menschliche Gesundheit besser zu verstehen. In diesem Projekt wurde untersucht, wie sich zukünftige Änderungen der Lebens- und Umweltbedingungen auf die Luftqualität in Innenräumen und der Außenluft auswirken werden. Berücksichtigt wurden u.a. Änderungen in Stadtplanung und Verkehrspolitik, demographischer Wandel, Klimawandel und Umweltpolitik. In einem zweiten Schritt wurden die Auswirkungen der Luftqualitätsänderungen auf die menschliche Gesundheit abgeschätzt. Das Projekt verfolgte einen interdisziplinären Ansatz unter Anwendung von modernsten Klimamodellen sowie Messungen, die Trends und mögliche Änderungen der Exposition beschreiben, und auf epidemiologische Expositions-Wirkungsfunktionen anwenden.

Innerhalb von ACCEPTED arbeitete die Universität Augsburg zusammen mit anderen Partnern hauptsächlich an der Entwicklung von Emissionsszenarien als Bestandteil von Luftqualitätsmodellierung für Augsburg sowie an der Wirkungsanalyse von Umweltzonen in den deutschen Städten Augsburg, München und Berlin. Darüber hinaus wurde ein umfangreicher Datensatz zur individuellen (personenspezifischen) Exposition (personal exposure) analysiert und zur Validierung eines Modells für die Abschätzung der personenspezifischen Luftschadstoffkonzentrationen bereitgestellt.

**Abstract**

This national research project was conducted in the framework of the transnational project ACCEPTED (Assessment of changing conditions, environmental policies, time-activities, exposure and disease). ACCEPTED involved 11 different partners from Belgium, Germany, France and Sweden and was funded by five organisations (ADEME and ANSES in France, BELSPO in Belgium, Swedish EPA in Sweden and UBA in Germany) within the framework of the ERA-ENVHEALTH network.

ACCEPTED is an international research project that aims to improve our understanding of future exposure situations in cities and their impact on health. This project investigated how the changes of living and environmental conditions in the future will affect the indoor and outdoor air quality. We considered among others changes in urban design and traffic policy, demography, climate and environmental policies. In a second step the impact of air quality changes on human health was estimated. This was achieved by using an interdisciplinary approach under the application of various state-of-the-art atmospheric models and measurements describing effects of trends and potential changes on exposure together with epidemiological exposure-response functions.

Within the ACCEPTED project the University of Augsburg was involved together with other project partners mainly in development of emission scenarios as part of the air quality modeling in Augsburg and in the evaluation of the effectiveness of Low Emission Zones (LEZ) implemented in three German cities: Augsburg, Munich and Berlin. In addition University Augsburg analyzed large data

sets of personal measurements and provided those data for validation of personal exposure modeling to air pollutants.

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# Abkürzungsverzeichnis

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|---|---|
| **ACCEPTED** | ERA-ENVHEALTH-Projekt "Assessment of changing conditions, environmental policies, time-activities, exposure and disease" |
| **ADEME** | The French Environment and Energy Management Agency |
| **ANSES** | French Agency for Food, Environmental and Occupational Health & Safety |
| **AIRPARIF** | Umweltbehörde in Paris: Surveillance de la qualité de l'air en Île-de-France (air quality monitoring network in Paris) |
| **AirViro** | Gaußmodell zur Bestimmung der verkehrsbedingten Belastung durch Straßennetze unter Nutzung von Immissions- und Emissionsdaten, sowie Dispersions- und Windmodellierung (http://www.airviro.smhi.se). |
| **AP** | Arbeitspakete des Projekts |
| **ArcGIS** | Oberbegriff für verschiedene GIS-Softwareprodukte des Unternehmens ESRI (Environmental Systems Research Institute) |
| **BC** | Schwarzer Kohlenstoff (kohlenstoffhaltige Partikel, Ruß, Black Carbon) |
| **BELSPO** | Belgian Science Policy |
| **BP** | Bourgesplatz (Augsburg) |
| **CPB-Modell** | Straßenschluchtmodell (Canyon Plume-Box Modell) |
| **CPC** | Kondensations-Partikel Zähler (Condensation Particle Counter) |
| **DTV** | durchschnittliche tägliche Verkehrsstärke (in Kfz/24h, Daily Traffic Volume) |
| **DWD** | Deutscher Wetterdienst |
| **EC** | Elementarer Kohlenstoff (Ruß als elementarer Kohlenstoff, Elemental Carbon) |
| **ERA-ENVHEALTH** | European Research Area Environment and Health |
| **HA** | Hochschule Augsburg |
| **GIS** | Geographisches Informationssystem |
| **IMMIS$^{luft}$** | Screening-Programm zur Bestimmung der Luftschadstoff-Emissionen und -Immissionen in Innenstädten (http://www.immis.de/) |
| **LfU** | Bayerisches Landesamt für Umwelt in Augsburg |
| **LÜB** | Lufthygienisches Landesüberwachungssystem Bayern |
| **$NO_2$** | Stickstoffdioxid |
| **$NO_x$** | Stickstoffoxide |
| **OC** | Organischer Kohlenstoffanteil (Organic Carbon) |

| | |
|---|---|
| **OM** | Organische Masse |
| **p/A** | Personenspezifische Konzentration/Außenluftkonzentration |
| **$PM_{10}$** | $PM_{10}$-Feinstaub (Partikel mit einem aerodynamischen Durchmesser kleiner als 10 Mikrometer) |
| **$PM_{2,5}$** | $PM_{2,5}$-Feinstaub (Partikel mit einem aerodynamischen Durchmesser kleiner als 2,5 Mikrometer) |
| **PNC** | Partikelanzahlkonzentration (particle number concentration) |
| **SMHI** | Schwedisches Institut für Meteorologie und Hydrologie in Stockholm (Swedish Meteorological and Hydrological Institute) |
| **Swedish EPA** | Swedish Environmental Protection Agency |
| **TC** | Gesamtkohlenstoff (Total Carbon) |
| **UBA** | Umweltbundesamt |
| **UFP** | ultrafeine Partikel (Partikel mit einem Durchmesser < 100nm) |
| **UWZ** | Umweltzone - im internationalen Sprachgebrauch oft als low emission zone (LEZ) bezeichnet |

## Zusammenfassung

ACCEPTED (Assessment of changing conditions, environmental policies, time-activities, exposure and disease) ist ein interdisziplinär angelegtes Forschungsprojekt mit dem Ziel, zukünftige Expositionsszenarien in Städten und ihren Einfluss auf die menschliche Gesundheit besser zu verstehen. Hierfür wurden modernste Klimamodelle sowie Messungen verwendet, die Trends und mögliche Änderungen der Exposition beschreiben, und auf epidemiologische Expositions-Wirkungsfunktionen anwenden.

Im Rahmen des Projekts wurden Methoden entwickelt, die eine Abschätzung der Maßnahmen zur Verbesserung der Luftqualität hinsichtlich ihrer Effektivität erlauben. Die Abschätzung erfolgte sowohl durch statistische Auswertung der vorhandenen Messreihen als auch durch Modellrechnungen. Die Analyse der Wirksamkeit der Umweltzone in den drei deutschen Städten Augsburg, München und Berlin stellte einen Schwerpunkt der an der Universität Augsburg durchgeführten Arbeiten dar. Darüber hinaus lag in der Auswertung bereits vorhandener Daten zur personenspezifischen Exposition durch $PM_{2,5}$ (Partikel, deren aerodynamischer Durchmesser kleiner als 2,5 Mikrometer ist), kohlenstoffhaltige Partikel (Black Carbon - BC) und ultrafeine Partikel (UFP) ein weiterer Beitrag der Universität Augsburg zum Gesamtprojekt.

Eine Umweltzone in Deutschland ist ein definiertes Gebiet (meist in städtischen Ballungsräumen) das nur den Zugang von Fahrzeugen erlaubt, die bestimmte Abgasnormen erfüllen. Die Fahrzeuge werden mit Plaketten (rot, gelb, grün) gekennzeichnet, die sich farblich an den EU-Abgasnormen (Euro 1 bis 6) orientieren. Auch in anderen EU-Staaten sind Umweltzonen (oder vergleichbare Maßnahmen wie zum Beispiel City-Maut) eingerichtet; es besteht jedoch keine einheitliche Regulierung der Umweltzonen in den unterschiedlichen EU Mitgliedstaaten.

Die Wirksamkeit von Umweltzonen wird in Modellrechnungen und zunehmend anhand von $PM_{10}$-Immissionsdaten untersucht. Wenngleich die Ergebnisse anfänglich aufgrund ungenauer Schätzungen bzw. zu kurzer Messreihen widersprüchlich ausfielen, zeigen neuere Analysen jedoch einen klaren Trend. So ist bei ausreichender Größe der Umweltzone und Geltung der strengsten Schadstoffgruppe (dritte Stufe, Einfahrt nur mit grüner Plakette) ein Rückgang der $PM_{10}$-Konzentrationen um 5-10% nachweisbar, an verkehrsbelasteten Messstationen teilweise auch um über 10% (Cyrys et al., 2014).

Auch die im Rahmen dieses Projekts mit einem statistischen Regressionsmodell durchgeführte Analyse der $PM_{10}$-Konzentrationen für die großen Umweltzonen in München und Berlin bestätigt diesen Trend. In München sind die Minderungseffekte nach der Verschärfung der Regelungen für die Umweltzone von Stufe 1 auf Stufe 2 an allen Messstationen statistisch signifikant. Insgesamt ist die Abnahme der $PM_{10}$-Belastung im Sommer größer als im Winter. Auch in Berlin fallen die Minderungseffekte nach der Verschärfung der Regularien (Stufe 3) deutlicher aus als nach der Einführung der ersten Stufe. Diese Reduktion ist besonders im Sommer sichtbar und schwächer ausgeprägt im Winter. Anscheinend tragen in den Wintermonaten Partikel aus anderen Quellen (wie zum Beispiel Hausbrand, Holzverbrennung, aufgewirbelter Staub bedingt durch die Anwendung von Streusalz) signifikant zu den $PM_{10}$-Konzentrationen im Winter bei und folglich ist eine Maßnahme, die nur Fahrzeugabgase reguliert, weniger effektiv. In Augsburg, wo die Umweltzone deutlich kleiner ist, waren die Ergebnisse der Wirkungsanalyse nicht konsistent, weder im Hinblick auf die Unterschiede zwischen Sommer und Winter noch im Hinblick auf die Unterschiede zwischen Verkehrsmessstationen und städtischen Hintergrundstationen. Eine statistisch signifikante Minderung der $PM_{10}$-Belastung wurde nur im Sommer an Verkehrsmessstationen beobachtet, nicht aber an Hintergrundmessstationen. Darüber hinaus sind keine Veränderungen der $PM_{10}$-Konzentrationen nach der Verschärfung der Regularien von Stufe 1 auf Stufe 2 auszumachen.

Folglich kamen wir zu dem Schluss, dass der Einfluss der Umweltzone in Augsburg auf die $PM_{10}$-Konzentrationen eher schwach, oder gar nicht vorhanden ist.

In Augsburg wurde zusätzlich zur Analyse der vorhanden $PM_{10}$ Immissionsdaten die Wirkung der Umweltzone auf die $PM_{10}$- und $NO_x$-Konzentrationen durch Modellierung abgeschätzt. Angewendet wurde das AirViro Modellierungssystem. AirViro ist ein Gaußmodell zur Bestimmung der verkehrsbedingten Belastung durch Straßennetze unter Nutzung von Immissions- und Emissionsdaten, sowie Dispersions- und Windmodellierung. Die Ergebnisse der Analyse der Wirksamkeit der Einführung der Umweltzone in Augsburg durch Modellierung lassen sich wie folgt zusammenfassen:

- $PM_{10}$: an einer Hintergrundmessstation im Stadtzentrum liegt der Beitrag der lokalen Quellen vom Stadtgebiet Augsburg zur Gesamtbelastung (ca. 17-18 µg/m³) bei etwa 10% (2 µg/m³). Der regionale Beitrag von Quellen außerhalb Augsburgs in der Größenordnung von etwa 15 µg/m³ macht die restlichen 90% der gemessen Konzentrationen im städtischen Hintergrund aus. Die jährliche Variation des regionalen Beitrages lag in der Größenordnung von etwa 3 µg/m³ für die Jahre 2008-2011.
- $NO_2$/$NO_x$: der Beitrag der lokalen Quellen vom Stadtgebiet Augsburg zur $NO_x$-Konzentration im städtischen Hintergrund beträgt im Mittel etwa 15 µg/m³ (19 µg/m³ am Bourgesplatz). Der regionale Hintergrund wird auf etwa 20 µg/m³ $NO_x$ geschätzt. Das bedeutet dass die lokalen Quellen bis zu 50% der gemessenen $NO_x$-Konzentrationen beitragen.
- Die erwartete Reduktion von städtischer Hintergrundkonzentration aufgrund der Einführung der Umweltzone in Augsburg liegt bei 0.3 µg/m³ für $PM_{10}$ und 4 µg/m³ für $NO_x$ für die Umweltzone in der dritten Stufe. Das bedeutet eine Reduktion der Jahresmittelwerte von $PM_{10}$ in der Größenordnung von 1-2% und von unter 10% für $NO_x$ im städtischen Hintergrund. Die Jahr-zu-Jahr Variation der Konzentration liegt bei 2.5 µg/m³ für $PM_{10}$ und bei 10 µg/m³ für $NO_x$

Die Ergebnisse der Dispersions-Modellierung zeigen deutlich, dass die lokalen Emissionen im Stadtgebiet Augsburg nur zur etwa 10% der gemessenen $PM_{10}$-Konzentrationen beitragen, während die lokalen Quellen zur etwa 30 bis 50% der gemessenen $NO_2$-Konzentrationen beitragen. Die erwartete Reduktion im städtischen Hintergrund aufgrund der Einführung der Umweltzone liegt für $PM_{10}$ zwischen 1 und 2% und für $NO_x$ bei 10%. Die Größenordnung der Jahr-zu-Jahr Variation von $PM_{10}$- und $NO_x$-Konzentrationen, die auf die Änderungen des meteorologischen Bedingungen zurückzuführen sind, ist viel größer als der erwartete Effekt der Umweltzone. Dies macht deutlich, dass die Analyse der Effekte von Umweltzonen anhand von gemessenen Daten nicht durch einen einfachen Vergleich der Jahresmittelwerte erfolgen kann. Stattdessen müssen anspruchsvolle statistische Analysen mit Adjustierung auf meteorologischen Bedingungen vorgenommen werden. Dieser Ansatz wurde in der vorliegenden Studie gewählt. Durch eine Adjustierung auf die Werte einer Referenzmessstation wurden der störende Einfluss der Meteorologie und der Langzeittrend des regionalen Beitrages zur $PM_{10}$-Konzentration in Augsburg herausgerechnet. Die Ergebnisse der AirViro Modellierung deuten aber darauf hin, dass der erwartete Rückgang der $PM_{10}$-Konzentration an allen städtischen Hintergrundmessstation exakt in der gleichen Größenordnung liegt, wie der erwartete Rückgang der $PM_{10}$-Konzentration an der Referenzmessstation im regionalen Hintergrund. Dies erklärt warum wir den Effekt der Umweltzone im städtischen Hintergrund in Augsburg durch Anwendung der statistischen Modellierung nicht nachweisen konnten. Beide Modellierungen zusammenfassend lässt sich Folgendes schlussfolgern: die statistische Modellierung konnte einen Effekt der Umweltzone im städtischen Hintergrund in Augsburg nicht nachweisen, während die Dispersions-Modellierung eine mögliche Erklärung dafür erkennen lässt.

Darüber hinaus zeigt ein Vergleich der Zusammensetzung der regionalen Fahrzeugflotte in Augsburg mit dem Bundesdurchschnitt, dass die Kfz-Fahrzeuge in Augsburg vor einigen Jahren deutlich moderner (d.h. mit geringeren Schadstoffemissionen) waren, als im Bundesdurchschnitt. Das bedeutet, dass alleine aus diesem Grund der Effekt der Umweltzone in Augsburg schwächer sein muss als zum Beispiel in Berlin, wo die Fahrzeugflotte erst aufgrund der Einführung der Umweltzone in den letzten Jahren deutlich erneuert wurde.

Die mit dem AirViro-Modell durchgeführte Abschätzung der Wirksamkeit der Umweltzone in Augsburg wurde mit einer Abschätzung verglichen, die mit einem deutschen Modellierungssystem $IMMIS^{luft}$ durchgeführt wurde. Der Vergleich zeigt, dass die durch beide Modelle geschätzten Effekte in der gleichen Größenordnung liegen. Die Reduktion von $PM_{10}$-Konzentrationen in Augsburg wurde durch das $IMMIS^{luft}$ Modell unter Ausblendung der allgemeinen Flottenverjüngung auf rund 0,3 µg/m³ geschätzt. Diese Abschätzung wurde für die Einführung der ersten Stufe der Umweltzone durchgeführt und bezog sich auf hochbelastete Straßenstandorte. AirViro prognostiziert eine Reduktion der abgasbedingten $PM_{10}$-Konzentrationen um 0,1 µg/m³ bei der Einführung der ersten Stufe und um bis zu 0,3 µg/m³ bei der Einführung der dritten Stufe der Umweltzone. Diese Analyse wurde für Standorte im städtischen Hintergrund durchgeführt.

Der maßgebliche Grund für die Einführung der strengeren $PM_{10}$-Grenzwerte ab 2005 ist das Wissen um die negativen gesundheitlichen Auswirkungen des Feinstaubs. Die gesundheitlich relevanteste $PM_{10}$ Partikelfraktion besteht aus verkehrsbedingten Partikeln und hier insbesondere aus Dieselruß-Partikeln. Deshalb fördern die deutschen Regularien der Umweltzonen die Nachrüstung von Dieselfahrzeugen mit Rußpartikeln-Filtern. Leider stützt sich die „Erfolgskontrolle“ nur auf die $PM_{10}$-Konzentration, die durchschnittlich nur ca. 20% der hochtoxischen Partikel aus KfZ-Verbrennungsmotoren repräsentiert. Eine Absenkung dieses Anteils auf ca. 10% geht mit einer Absenkung des Rußanteils auf ca. 50% einher, wie dies in „leistungsfähigen“ Umweltzonen gezeigt werden konnte. Das bedeutet gleichzeitig, dass der Vorteil der Umweltzonen für die Gesundheit erheblich größer sein kann, als es sich allein an der Reduktion von $PM_{10}$-Konzentration ablesen lässt.

Berlin ist eine der wenigen europäischen Städten in denen die Gesamtkohlenstoff-Konzentration (total carbon, TC) an Hauptverkehrsstraßen seit den 90-er Jahren gemessen wird. Im Rahmen dieses Projekts wurde zusätzlich eine Analyse dieser Daten durchgeführt. Die Abnahme der TC-Konzentration fiel deutlich stärker aus als die entsprechende $PM_{10}$-Minderung. Während die $PM_{10}$-Konzentration an einer Verkehrsmessstation nach der Einführung der Umweltzone um 16% im Sommer und um 9% im Winter abgenommen hat, ging die dazugehörige TC-Konzentration um 24% (im Sommer) und 16% (im Winter) zurück. Ähnliche Ergebnisse wurden aus Leipzig und München berichtet. So ging in München nach der Einführung der Umweltzone die mittlere EC-Konzentration im Faktor „Kfz-Verkehr“ von 1,1 (vor der Einführung der Umweltzone) auf 0,5 µg/m³ zurück (nach der Einführung der Umweltzone), was eine Abnahme um 50% bedeutet (Qadir et al., 2013). Löschau und Kollegen (2013) berichten für Leipzig über eine relative Minderung von EC-Konzentrationen um 24% und von Ruß-Konzentrationen (gemessen als black carbon, BC) um 31% an einer Messstation im Stadtzentrum nach der Einführung der Umweltzone. Dieser Rückgang scheint in erster Linie ein Effekt der Abnahme der Verkehrsdichten in der Umweltzone zu sein, und erst in zweiter Linie ein Effekt der Reduktion von Abgasemissionen. Für $PM_{10}$-Konzentration konnte in Leipzig dagegen keine Änderung beobachtet werden.

Insgesamt zeigen die Ergebnisse, dass sich die Umweltzonen als erfolgreiche Maßnahme für die Luftreinhaltung erweisen, wenn sie groß genug sind und möglichst wenige Ausnahmen zulassen (wie in Berlin und München). Sie senken nicht nur $PM_{10}$-Konzentration, sondern in einem viel größeren Ausmaß auch die gesundheitlich relevanten Komponenten (wie zum Beispiel Dieselruß), die im $PM_{10}$ enthalten sind. Folglich kann der Nutzen der Umweltzonen viel besser abgeschätzt werden, wenn neben $PM_{10}$ weitere Messparameter, wie Ruß (BC) und elementarer Kohlenstoff (EC), gemessen werden.

Die Auswertung der personenspezifischen Messungen hat gezeigt, dass die individuelle Exposition zu $PM_{2,5}$, BC und UFP immer höher waren, als entsprechende Konzentrationen dieser Schadstoffe in der Außenluft. Diese Erhöhung ist besonders deutlich ausgeprägt für verkehrsbezogene Aufenthaltsorte und weniger deutlich in reinen Wohngebieten (im städtischen Hintergrund). Die verkehrsabhängige BC- und UFP-Konzentrationen waren sehr unterschiedlich in den unterschiedlichen Szenarien, sowohl bezüglich der absoluten Konzentrationen der Schadstoffe als auch im Hinblick auf das Verhältnis zwischen der **p**ersonenspezfischen (individuellen) Exposition und der Konzentration in der **A**ußenluft (p/A Verhältnis). Im Gegensatz dazu waren die Unterschiede für $PM_{2,5}$-Konzentrationen in den unterschiedlichen Szenarien weniger deutlich.

Die Korrelation zwischen der individuellen Konzentration und der Außenluftkonzentration für die zeitlich hoch aufgelösten Messungen (in minütlicher oder stündlicher Auflösung) waren deutlich niedriger als die gleichen Korrelationskoeffizienten für die Tagesmittelwerte. Das bedeutet, dass die tägliche Variation der Luftschadstoff-Konzentration in der Außenluft (gemessen im städtischen Hintergrund) durchaus einen signifikanten Einfluss auf die personenspezifische Schadstoffkonzentration hat, insbesondere für Personen die sich im Freien aufhalten. Diese Ergebnisse sind hoch relevant für die Interpretation der Zusammenhänge zwischen Innenraum-, Außenluft- und individueller Exposition und den daraus folgenden Konsequenzen für epidemiologische Kurzzeitstudien. Im Allgemeinen ist die Korrelationen zwischen personenspezifischer Exposition und Außenluftkonzentration schwach (Wallace and Williams, 2005). Allerdings ist für epidemiologische Studien nicht die Korrelation zwischen der gesamten personenspezifischen Exposition und der Außenluft auschlaggebend, sondern die Korrelation zwischen den Komponenten der personenspezifischen Exposition, die von der Außenluft stammen, und der Außenluftkonzentration. Deswegen sollte in epidemiologischen Studien, die die Außenluftkonzentration als Surrogat für Exposition verwenden, die gesamte personenspezifische Exposition in die Beiträge der Außenluft und des Innenraums getrennt werden. Nur so kann untersucht werde, ob die Außenluftkonzentration stellvertretend für Außenluft-Exposition verwendet werden kann. In unserer Studie war der Einfluss der Innenraumquellen auf die individuelle Exposition vernachlässigbar. Die in der Studie beobachteten hohe Korrelation zwischen den Tagesmittelwerten der Außenluftkonzentration und der individuellen Exposition deuten darauf hin, dass die täglichen Schwankungen der Beiträge der Außenluft zur personenspezifischen Exposition durchaus durch Messungen in der Außenluft an einer stationären Messstation im städtischen Hintergrund abgeschätzt werden können. Die schwächeren Korrelationen zwischen der gesamten personenspezifischen Exposition und der Außenluftkonzentration von Schadstoffen, die in anderen Studien beobachtet wurden, ist auf den störenden Einfluss der Innenraumquellen zurückzuführen, der von menschlichen Tätigkeiten abhängig ist und deshalb von Person zu Person stark variiert. Es scheint, dass die Anteile der Außen- und der Innenraumluft, die zur gesamten personenspezifischen Exposition beitragen, kaum korreliert sind. So gesehen können die Außenluft- und Innenraumquellen als unabhängig betrachtet werden. Folglich kann der Zusammenhang zwischen Innenraum- und Außenluftschadstoffen separat untersucht werden. So kann man die zunächst paradox erscheinende Tatsache verstehen, dass statistisch signifikante Zusammenhänge zwischen Gesundheitseffekten und Außenluftkonzentration gefunden werden, obwohl sich Menschen im Durchschnitt zu 90% in Innenräumen aufhalten und die Korrelationen zwischen Außenluft- und Innenraumkonzentration von Luftschadstoffen sehr schwach sind. Die gemessene Außenluftkonzentration ist offensichtlich ein guter Indikator für den Außenluftanteil der personenspezifischen Exposition.

## Summary

ACCEPTED (Assessment of changing conditions, environmental policies, time-activities, exposure and disease) is a research project that aims to improve our understanding of future exposure situations in cities and their impact on health, from an interdisciplinary approach. This is achieved by using various state-of-the-art atmospheric models and measurements describing effects of trends and potential changes on exposure together with epidemiological exposure-response functions.

In the framework of this project we have developed methods for evaluation the effectiveness of measures aiming air quality improvement. The evaluation was based on statistical analysis of existing monitoring data as well modeling. The focus of the work conducted by the University of Augsburg was on the analysis of the effectiveness of Low Emission Zones (LEZs) implemented in the three German cities Augsburg, Berlin and Munich. The analysis of data on personal exposure to $PM_{2.5}$ (particulate matter with an aerodynamic diameter < 2.5 µm), black carbon (BC) and ultrafine particles (UFP) was further contribution of the University of Augsburg to the whole project.

A LEZ in Germany is a defined area (mostly in urban agglomerations) where vehicles have to meet certain emissions standards when entering it. The vehicles are identified by windscreen badges that come in a colored code (red, yellow, green) which is directly linked to the corresponding stages of European emission standards (Euro 1 to 6). LEZs (or other measures such as Congestion Charge Zones) have been implemented also in other European countries; however there are no uniform regulations for LEZs in the different EU member states.

The effectiveness of LEZs is investigated by dispersion modeling or increasingly also by analysis of $PM_{10}$ measurement values. While initially inconsistent results were reported due to inaccurate estimates or short time series of $PM_{10}$ measurements, recent studies show a clear trend. In a sufficiently large area coupled with the application of the strictest emission standards (third stage, i.e. entry only with green badge) a reduction of $PM_{10}$ concentration between 5 and 10 % can be observed; at traffic site occasionally above 10 % (Cyrys et al., 2014).

The analysis of the $PM_{10}$ concentration changes after the implementation of the large LEZs in Munich and Berlin, which was conducted in the framework of this project using a statistical regression model, confirms these findings. In Munich we observed a statistically significant decrease of $PM_{10}$ mass concentration at all monitoring sites after introducing stricter regulations from LEZ stage 1 to LEZ stage 2. Also in Berlin the decrease after the implementation of the LEZ stage 3 was larger compared to the reduction after implementation of the LEZ stage 1. This reduction of $PM_{10}$ levels is in general more visible in the summer season and less pronounced for the winter season. Apparently particles from other sources (such as domestic heating, wood combustion, resuspended dust due to the application of road salt for deicing) contribute significantly to the $PM_{10}$ mass concentrations in winter and consequently a measure regulating only vehicle exhaust particles becomes less effective.

The results for Augsburg, with a considerable smaller LEZ, were not consistent both regarding the differences between summer and winter season as well as between traffic sites and urban background sites. A statistically significant decrease of $PM_{10}$ concentrations was observed only in the summer season and only at traffic monitoring sites, and not in the urban background. Furthermore, the $PM_{10}$ levels in Augsburg remained stable after the implementation of the second stage of the LEZ. Therefore we concluded that the impact of the LEZ in Augsburg on the $PM_{10}$ levels is either weak or non-existent.

In Augsburg, we have estimated the effects of the LEZ introduction on $PM_{10}$ and $NO_2/NO_x$ levels by using the Airviro dispersion model in addition to the analysis of the $PM_{10}$ monitoring data. Airviro

is a Gaussian dispersion model for the estimation of the traffic related air pollution levels due to road network which comprises measurement data, emission inventories, dispersion and wind modelling tools. The results of the estimation the effectiveness of the LEZ in Augsburg by modelling can be summarized as follows:

- $PM_{10}$: Local sources within the city area of Augsburg contribute to about 10% (2 µg/m³) of monitored levels (about 17-18 µg/m³) at an urban background site in the city center. Regional contribution from sources outside of Augsburg of about 15 µg/m³ explains the other 90% of the monitored levels in the urban background. The annual variation of this regional contribution was approximately 3 µg/m³ during the period 2008-2010.
- $NO_2/NO_x$: The contribution of the local sources within the modelling area was about 15 µg/m³ of $NO_x$ on average in the urban background (19 µg/m³ of $NO_x$ at Bourgesplatz site). The regional background contribution is estimated to be about 20 µg/m³ of $NO_x$. It means that the local sources contribute up to 50% of the monitored $NO_x$ levels.
- The expected reductions of the urban background levels due to the LEZ implementation in Augsburg are up to 0.3 µg/m³ for $PM_{10}$ and 4 µg/m³ for $NO_x$ for the LEZ in stage 3. It means a reduction of the $PM_{10}$ annual average concentration approximately 1-2% and of slightly under 10% for $NO_2$ in the urban background. The year-to-year concentration variation is 2.5 µg/m³ for $PM_{10}$ and 10 µg/m³ for $NO_x$.

The dispersion modelling results show clearly that emissions within Augsburg contribute only to about 10% of monitored $PM_{10}$ levels, while local emissions contribute to about 30 – 50% of monitored $NO_2$ levels. The implementation of the LEZ is expected to result in 1-2% lower $PM_{10}$ concentration and slightly below 10% lower $NO_x$ concentration in the urban background. The magnitude of the year-to-year variation in $PM_{10}$ and $NO_x$ levels due to meteorological conditions is much larger than the expected impact of the LEZ. This shows that the analysis of the LEZ effect using monitoring data could not be done by simple comparison of annual averages of those pollutants. Instead, sophisticated statistical analyses with adjustment for meteorological conditions need to be used. This approach was used in our study. We adjusted for meteorological conditions and long term temporal trends in the regional contribution to $PM_{10}$ levels using $PM_{10}$ levels measured at a reference station. However, the results of the AirViro dispersion modeling indicated that the expected reduction of $PM_{10}$ levels at all urban sites are exactly in the same range as the reduction at the reference site located in regional background. This explains why we were not able to detect any effects of the LEZ at background sites in Augsburg using the statistical modeling. Linking the two assessments: the statistical modelling could not find any significant effects of LEZ in the Augsburg urban background $PM_{10}$ data, whereas the dispersion modelling approach indicates a possible explanation for this.

Additionally, a few years ago in comparison to the national average, cars in use in Augsburg were significantly more modern (i.e. generating lower pollutant emissions) For this reason alone the effects of the LEZ in Augsburg should be weaker than for example in Berlin, where the cars were dramatically renewed in recent years due to the implementation of the LEZ.

The estimation of the effectiveness of the LEZ in Augsburg performed by the AirViro modeling system was compared to the estimation conducted by the German IMMIS$^{luft}$ dispersion model. The comparison shows that the effects estimated by both modeling systems were in the same range. The reduction of the $PM_{10}$ levels in Augsburg was estimated to be 0.3 µg/m³ by the IMMIS$^{luft}$ model not taking the renewal of the car fleet under consideration. This estimation was conducted for the LEZ in stage 1 and was related to highly polluted traffic sites. The reduction of the $PM_{10}$ levels due to exhaust emission was estimated to be 0.1 µg/m³ by AirViro after implementation of LEZ in stage 1, and up to 0.3 µg/m³ after the implementation of the LEZ in stage 3. This estimation was conducted for urban background locations.

The main reason for the implementation of limit values for $PM_{10}$ in 2005 was the knowledge about the adverse health effects of fine particles. The most health-relevant $PM_{10}$ particle fraction consists mainly of traffic related particles and here especially of diesel soot particles. Therefore, the German regulations for LEZs promote the use of diesel particulate filter (DPF) in diesel cars. Unfortunately, the "verification of success" of the LEZ is mostly restricted to $PM_{10}$, a particle fraction containing on average only 20% of toxic exhaust related particles. A 10% reduction of $PM_{10}$ leads to a reduction of the toxic and health-relevant diesel soot fraction of 50%, which has already been reported for some "efficient" LEZs. This means that the benefit of LEZs on human health is far greater than is presently visible from routine measurements of $PM_{10}$.

Berlin is one of the few European cities where total carbon (TC) concentrations have been measured at major roads since the1990s. Within the framework of this project we analyzed additionally those data. The decrease of TC concentrations was clearly larger than the corresponding decrease of $PM_{10}$ levels. Whereas $PM_{10}$ concentrations at a traffic site decreased after the implementation of the LEZ by 16% in summer and 9% in winter, the corresponding reduction of TC was 24% (summer season) and 16% (winter season). Similar results were reported from Leipzig and Munich. In Munich the average concentration of EC from traffic factor decreased from 1.1 (before the implementation of the LEZ) to 0.5 µg/m³ (after the implementation of the LEZ), which equals a 50% reduction (Quadir et al., 2013). Löschau et al. (2013) showed a relative reduction of EC concentration by 24% and soot (measured as black carbon, BC) concentration by 31% at a traffic monitoring site in the city center of Leipzig after implementation of the LEZ. This decrease seems to be firstly an effect of decreasing traffic volumes in the area of the LEZ and only secondly an effect of reductions in the vehicles' exhausts emissions. For $PM_{10}$ mass concentrations, no decrease in Leipzig could be observed.

Overall, the results show that LEZs are proving successful as a measure for air pollution control if they are large enough and only few exemptions are granted (as in Berlin and Munich). They decrease not only $PM_{10}$ but, to a much higher degree, the health-relevant components (such as diesel soot) contained in $PM_{10}$. Therefore, the benefit of the LEZs can be estimated much better if, in addition to $PM_{10}$, also diesel soot (as BC) and elemental carbon (EC) are measured.

The analysis of the personal monitoring data shows that personal exposure to $PM_{2.5}$, BC and UFP is always higher than the ambient concentration of these pollutants measured in the urban background by a stationary monitoring station. This increment is particularly pronounced for the measurements conducted in traffic related microenvironments and less pronounced for measurements conducted in residential areas (urban background). The traffic related BC and PNC levels were highly variable across different scenarios, both regarding the absolute levels of pollutants as well as the ratios between the personal exposure and the urban background concentrations. In contrast, $PM_{2.5}$ was found less variable in different scenarios.

The correlation between the personal exposure and the ambient air pollutant concentrations recorded in high temporal resolution (1-minute or hourly resolution) was clearly weaker than the same correlation for the daily means. This implicates that the day-to-day variation of the air pollutants in ambient air (measured in urban background) has a significant impact on the personal exposure, especially for persons being outdoors. The obtained results are highly relevant for the interpretation of the relationship between indoor, outdoor and personal exposure and the resulting consequences for epidemiological short-term studies. In general, the correlations between the personal exposure and outdoor concentration are weak (Wallace and Williams, 2005). However, the correlation of interest in epidemiological studies is not that between total personal exposure and outdoor concentrations, but the correlation between the component of personal exposure that is due to outdoor pollutants and the outdoor (ambient) concentrations. Therefore, in epidemiological studies that rely on ambient concentrations as a surrogate for exposure, total personal exposure must be divided into its ambient and indoor components to investigate whether ambient concentrations can be used as a surrogate for ambient exposure. In our study the impact of indoor sources to personal exposure was negligible. The strong correlations between the daily means of ambient concentration and the personal exposure observed in our study indicate that the day-to-day variation of the ambient component of personal exposure might be sufficiently reflected by a stationary measurement station in an urban background. The weaker correlations between total personal exposure and ambient pollutant levels observed in other studies are more likely caused by the indoor component of personal exposure, which depends more on human activity and varies from person to person. It seems that the ambient and indoor components of total personal exposure are hardly correlated. Accordingly, the relationship between ambient and indoor air pollutant sources can be considered as independent. Thus, the effect of indoor and outdoor air on human health can be estimated separately.

In this context it becomes clear what might seem paradoxical at first sight, namely that statistical significant associations between health effects and levels of ambient air pollutants have been observed, although humans spend on average 90% of their time indoors and the correlation between outdoor and indoor levels of air pollutants are very weak. The measured outdoor concentrations represent obviously good proxies for the ambient component of personal exposure.

# 1 Einleitung

Das Vorhaben wurde im Rahmen des transnationalen Forschungsprojekts ACCEPTED (Assessment of changing conditions, environmental policies, time-activities, exposure and disease) durchgeführt, an dem sich insgesamt 11 Partner aus Belgien, Deutschland, Frankreich und Schweden beteiligten. Das Projekt wurde von fünf Organisationen (ADEME und ANSES in Frankreich, BELSPO in Belgien, Swedish EPA in Schweden und UBA in Deutschland) im Rahmen des ERA-ENVHEALTH Netzwerks finanziert. In diesem Vorhaben sollte untersucht werden, wie sich zukünftige Änderungen der Lebens- und Umweltbedingungen auf die Luftqualität in Innenräumen und in der Außenluft auswirken werden. Es sollten u.a. der demographische Wandel, geänderte Lebensbedingungen in der Stadt, das Mobilitätsverhalten und der Klimawandel berücksichtigt werden. In einem zweiten Schritt wurden mit der Methoden der Gesundheitsfolgenabschätzung (Health Impact Assessment) die Auswirkungen der Luftqualitätsänderungen auf die menschliche Gesundheit abgeschätzt. Das Projekt verfolgte einen interdisziplinären Ansatz unter Anwendung von Modellierungsmethoden, Messungen und epidemiologischen Studien.

Das Projekt wurde in vier Arbeitspakete (AP) unterteilt: Luftqualität und Klima in Städten (AP1), personenspezifische Expositionsabschätzung in der Außenluft und im Innenraum (AP2), epidemiologische Studien an besonders gefährdeten Personen (AP3) sowie szenariobasierte Gesundheitsfolgenabschätzungen (AP4).

Im AP1 wurden Methoden entwickelt, die eine Abschätzung der Maßnahmen zur Verbesserung der Luftqualität hinsichtlich ihrer Effektivität erlauben. Diese Abschätzung erfolgte sowohl anhand von Modellrechnungen als auch durch die statistische Auswertung der vorhandenen Messreihen. Die Beurteilung der Effektivität der Einführung von Umweltzonen auf die Veränderungen der Luftqualität stellte einen Schwerpunkt dieses Arbeitspakets dar. Die Analyse war für Städte in Deutschland, Frankreich und Schweden geplant. Die Universität Augsburg entwickelte das statistische Regressionsmodell für diese Analyse und wendete das Modell auf die drei deutsche Städte Berlin, München und Augsburg an. Die Ergebnisse dieser Rechnungen erlauben es, Empfehlungen hinsichtlich der Größe der Umweltzonen, der erforderlichen Stufe der Regelung sowie der besten Strategie zur Überprüfung der Wirksamkeit von eingeführten Umweltzonen auszusprechen.

Die Wirksamkeit der Umweltzone in Augsburg sollte darüber hinaus auf Basis von Modellrechnungen mit dem Modellierungssystem AirViro analysiert werden. Die Modellierung wurde federführend vom Schwedischen Institut für Meteorologie und Hydrologie in Stockholm (Swedish Meteorological and Hydrological Institute, SMHI) durchgeführt. Die Universität Augsburg war durch die Beschaffung, Bereitstellung und Interpretation der lokalen Daten für Augsburg (Verkehrsplanung, Entwicklung von Luftreinhalteplänen, Stadtplanung- und Entwicklung, Immissionsmessdaten, meteorologische Daten) auch an der Entwicklung des Modellierungssystems AirViro beteiligt.

Darüber hinaus wurde die Modellierung der räumlichen Verteilung der Luftschadstoffkonzentrationen in Augsburg durch Anwendung eines in Deutschland entwickelten und bereits in mehreren Kommunen angewendeten Ausbreitungsmodells ($IMMIS^{luft}$) parallel durchgeführt. Die Ergebnisse der unterschiedlichen Ansätze der Modellrechnungen wurden miteinander verglichen und bewertet.

Im Arbeitspaket 2 bestand der Beitrag der Universität Augsburg in der Auswertung bereits vorhandener Daten zur personenspezifischen Exposition durch $PM_{2,5}$, kohlenstoffhaltige Partikel (TC) und ultrafeine Partikel (UFP). Die personenspezifischen Messungen wurden an unterschiedlichen Aufenthaltsorten durchgeführt. Diese Daten wurden anschließend dem Projektpartner AIRPARIF (Surveillance de la qualité de l'air en Île-de-France (air quality monitoring network), Paris) zur

Verfügung gestellt für die Validierung eines Modells zur Abschätzung der personenspezifischen Luftschadstoffkonzentrationen unter Einbeziehung der Konzentrationen in der Außenluft, in Innenräumen und personenspezifischen Tätigkeitsmustern.

Im Arbeitspaket 4 wurden die Auswirkungen der untersuchten Maßnahmen und Änderungen (z.B. Klimaänderungen) auf die Gesundheit abgeschätzt (Gesundheitsfolgenabschätzung). Die Universität Augsburg sollte die Gesundheitsfolgenabschätzung für die Einführung der Umweltzone in Augsburg durchführen.

In diesem Abschlussbericht sind die Ergebnisse der an der Universität Augsburg durchgeführten Analysen beschrieben, die im Rahmen des UFOPLAN-Vorhabens „Erarbeitung von Handlungskonzepten für eine gesunde Umwelt in städtischen Regionen unter Berücksichtigung des demographischen Wandels“ (FKZ 3712 61 20) finanziert wurden. In Kapitel 2 dieses Berichts sind die Ergebnisse der Wirkungsanalyse der Umweltzonen für die drei deutschen Städte Augsburg, Berlin und München durch statistische Analyse der Immissionsmessdaten zusammengefasst. In Kapitel 3 sind die durch den Einsatz von Modellierung erzielten Ergebnisse der Wirkungsanalyse der Umweltzone in Augsburg dargestellt. In Kapitel 4 sind die Ergebnisse der Analyse von personenspezifischen Expositionsmessungen zusammengefasst. Die Ergebnisse des gesamten ACCEPTED Projekts sind in englischer Sprache in einem separaten Bericht zusammengefasst. In Anlage IA und IB ist die deutsche und englische Zusammenfassung des Berichts zu finden. Der komplette Endbericht ist in Anlage II sowie auf der Internetseite des ACCEPTED Projekts zu finden (http://www.acceptedera.eu) und wurde zusammen mit diesem Bericht in elektronischer Form eingereicht (Dateiname: ACCEPTED Final Report.pdf).

# 2 Wirkungsanalyse von Umweltzonen für Augsburg, München und Berlin anhand von Immissionsmessdaten

## 2.1 Was sind Umweltzonen und wie kann die Wirkung dieser Maßnahme validiert werden?

Eine Umweltzone (UWZ) – im internationalen Sprachgebrauch oft als low emission zone (LEZ) bezeichnet – ist eine europaweite Form kommunaler Maßnahmen, die zur Reduzierung der verkehrsbedingten Luftbelastungen in der Außenluft führen soll. Die Umweltzonen wurden eingeführt, weil in vielen größeren Städten die europäischen Luftqualitätsgrenzwerte für $PM_{10}$-Feinstaub und Stickstoffdioxid ($NO_2$) überschritten wurden.

Der $PM_{10}$-Feinstaub ist eine Unterfraktion des Schwebstaubs, für den in der englischsprachigen Literatur der Begriff „Particulate Matter – PM" verwendet wird. Im Folgenden werden die Schwebstaubpartikel als Partikel oder PM bezeichnet. Die Unterteilung des Schwebstaubs (Feinstaubs) in so genannte Fraktionen erfolgt meist anhand der Partikelgröße. Unter $PM_{10}$-Feinstaub fallen alle Partikel, deren aerodynamischer Durchmesser kleiner als 10 Mikrometer ist. Eine Teilmenge des $PM_{10}$-Feinstaubs stellen feinere Teilchen mit einem aerodynamischer Durchmesser von weniger als 2,5 Mikrometer dar, die als $PM_{2,5}$-Feinstaub bezeichnet werden. Im Gegensatz dazu wird der Größenbereich von 2,5 bis 10 Mikrometer als "Grobfraktion" des Feinstaubs oder als „coarse particle fraction" bezeichnet.

In Deutschland werden in innerstädtischen Gebieten und Ballungsräumen mit hohem Verkehrsaufkommen die EU-weit gültigen Grenzwerte für $PM_{10}$-Feinstaub und Stickstoffdioxid ($NO_2$) oft nicht eingehalten. Für solche Gebiete und Ballungsräume müssen Luftreinhaltepläne aufgestellt werden, die die erforderlichen Maßnahmen zur dauerhaften Verbesserung der Luftqualität festlegen. In solchen Plänen können die Umweltzonen (UWZ) als „Maßnahme" rechtlich verankert werden. Das Umweltbundesamt (UBA) stellt die von den Ländern und Kommunen gemeldeten Informationen über die Umweltzonen in einer Übersicht für das gesamte Gebiet der Bundesrepublik zusammen (http://gis.uba.de/website/umweltzonen/umweltzonen.php). In der aktuellen Übersicht sind 53 Umweltzonen als aktiv in Stufe 3 (Einfahrt nur für Kraftfahrzeuge mit einer grünen Plakette erlaubt), und lediglich eine als aktiv in Stufe 2 (grüne und gelbe Plakette erlaubt) ausgewiesen. Diese befindet sich in Neu-Ulm (Bayern). (Stand Juni 2016).

Als Umweltzone ist in Deutschland ein Gebiet definiert, in dem nur Kraftfahrzeuge fahren dürfen, die bestimmte EU-Abgasstandards einhalten. Fahrzeuge mit besonders hohen Emissionen dürfen nicht in die Umweltzone hineinfahren. Um Fahrzeuge nach ihrer Schadstoffklasse unterscheiden zu können, werden sie in vier verschiedenen Schadstoffgruppen eingeordnet und die Einteilung der Fahrzeuge wird mit farbkodierten Plaketten visuell gekennzeichnet (keine Plakette, rote, gelbe und grüne Plakette). Die Einteilung der Schadstoffgruppen richtet sich nach den Emissionsschlüsselnummern der einzelnen Fahrzeuge. In eine Umweltzone sind nur die Fahrzeuge einfahrberechtigt, die über eine entsprechende Plakette verfügen. Das Befahren einer Umweltzone ohne entsprechend gültige Plakette kann als Ordnungswidrigkeit mit einem Bußgeld geahndet werden. Allerdings werden für bestimmte Fahrzeuggruppen (Arbeitsmaschinen, land- und forstwirtschaftliche Zugmaschinen, Krankenwagen, Oldtimer u.a.) Ausnahmeregelungen erteilt.

Alle Benzinfahrzeuge mit einem geregelten Katalysator bekommen eine grüne Plakette. Die Zuteilung der Plaketten für Dieselfahrzeuge ist stärker von den Schadstoffgruppen abhängig (keine Plakette für Euro 1, rote Plakette für Euro 2, gelbe Plakette für Euro 3 und schließlich grüne Plakette für Euro 4). Durch Nachrüstung eines Dieselfahrzeuges mit einem Partikelminderungssystem, beispielsweise Partikelfilter, kann die Eingruppierung in eine höhere Schadstoffgruppe und somit die Zuteilung einer gelben oder sogar einer grünen Plakette erreicht werden.

Die Schärfe der Regelungen kann in den einzelnen Umweltzonen durch die jeweilige Kommune bestimmt werden. So dürfen bei der ersten Stufe der Umweltzone Fahrzeuge ohne Plakette nicht mehr einfahren. Bei eingerichteter Stufe 2 oder 3 sind dann auch Fahrzeuge mit roter oder gelber Plakette betroffen und dürfen in das als Umweltzone ausgewiesene Gebiet nicht mehr einfahren.

In Deutschland befinden sich fast alle derzeit aktiven Umweltzonen in Stufe 3, sind also hinsichtlich der Schärfe der Regelungen sehr gut vergleichbar. Allerdings unterscheiden sich die Umweltzonen sehr stark hinsichtlich ihrer Größe, der zeitlichen Abstufung der Regelungen und zusätzlichen Maßnahmen (wie Lkw-Durchfahrtsverbot).

Die Minderungswirkung der Umweltzone wurde in vielen Kommunen vor der Einführung dieser Maßnahme durch Modellrechnungen und anhand von aktuellen Verkehrsdaten prognostiziert. Die ortspezifische Änderung der Schadstoffklassen wurde ebenfalls berücksichtigt. Bei diesen Untersuchungen geht man davon aus, dass die Einführung der Umweltzone zu einem schnelleren Austausch älterer Fahrzeuge führt. Dies bedeutet, dass sich die Wirkung der Umweltzone auf das gesamte Straßennetz einer Stadt auswirkt. Eine Schätzung des Umweltbundesamtes, die vor der Einführung der Umweltzonen durchgeführt wurde, prognostizierte Immissionsminderungen von $PM_{10}$-Feinstaubkonzentrationen von bis zu 10%, je nach technischem Zustand der Fahrzeugflotte (Diegmann et al. 2006).

Auch in einer Wirkungsanalyse der Landeshauptstadt München (LRP 2010) ist eine Minderung von $PM_{10}$-Konzentrationen aufgrund der Einführung einer Umweltzone in der Stufe 3 zwischen 2% und 10% prognostiziert. Die größte Minderung ist an stark verkehrsbelasteten Abschnitten des Mittleren Ringes zu erwarten. Abschätzungen aus anderen Städten haben ähnliche Ergebnisse gezeigt. Das Umweltbundesamt (UBA) hat die Ergebnisse vieler Modellrechnungen zusammengetragen und prognostizierte für die Stufe 1 der Umweltzone eine etwa 2 % Verminderung des $PM_{10}$-Feinstaubs (bezogen auf den Jahresmittelwert) (UBA, 2008). In Stufe 3 sind Verminderungen des $PM_{10}$-Feinstaubs um bis zu 10% möglich.

Diese relativ geringe Minderung der $PM_{10}$-Konzentrationen, die im Bereich von 2 bis 10 % liegt, ist der Hauptkritikpunkt an der Einführung von Umweltzonen. Es wird kritisiert, dass die Einrichtung der Umweltzonen zwar mit der Feinstaubproblematik begründet wird, aber nach vielen Abschätzungen die Reduktion der Feinstaubkonzentration nur wenige Prozent ausmacht. Darüber hinaus wird diskutiert, ob diese Veränderungen wegen des störenden Einflusses der Meteorologie messtechnisch überhaupt nachzuweisen sind.

Morfeld und Kollegen (2011) haben für die Ermittlung der Konzentrationsänderungen von Schadstoffen durch die Einführung von Umweltzonen die Anwendung von Regressionsmodellen empfohlen. Die Regressionsmodelle sollen nicht nur auf die Werte der Referenzmessstation adjustiert werden, sondern auch auf weitere wichtige Einflussfaktoren, wie zum Beispiel Mischungsschichthöhe, Windgeschwindigkeit oder Niederschlagsmenge. Die Autoren betonen, dass bei der geringen Größe des zu erwartenden Effektes die Wahl eines statistisch tragfähigen methodischen Ansatzes mit Berücksichtigung von Störfaktoren (Confounder) entscheidend ist, um Fehlinterpretationen zu vermeiden.

Für das von Morfeld und Kollegen (2011) ausgewählte Regressionsmodell werden Messwert-Quadrupel gebildet, die die zentrale Beobachtungseinheit bilden. Diese Quadrupel bestehen aus vier einander zugeordneten Messwerten von $PM_{10}$: ein Messwertpaar aus Index- und zeitgleichem Referenzmesswert liegt in der Beobachtungsphase II (Umweltzone aktiv), das andere Paar aus einem anderen Index- und zeitgleichem Referenzmesswert liegt in Beobachtungsphase I (Umweltzone nicht aktiv). Diese Paare haben einen zeitlichen Abstand von 364 Tagen (oder Vielfachen von 364 Tagen). In einer Pilotstudie haben Morfeld und Kollegen (2013) dieses Modell auf die gleichen Zeitperioden (Oktober 2007 bis Januar 2008 als Referenzmessperiode vor der Einführung der Umweltzone und Oktober 2008 bis Januar 2009 für die Periode nach der Einführung der Umweltzone) wie bei Cyrys und Kollegen (2009) angewendet. Insgesamt waren die Ergebnisse sehr heterogen mit Effektschätzungen von -4,0 % am Stachus und -2,7 % $PM_{10}$ an der Landshuter Allee (Verkehrsmessstation) bis +8 % an der Lothstrasse im städtischen Hintergrund. Für die anderen Messstationen (Prinzregentenstrasse, Luise-Kiesselbach-Platz) waren die Effektschätzer nicht signifikant. Die starke Zunahme der PM10-Konzentrationen an der Lothstrasse kann nicht mit Änderungen der Emissionen aufgrund der Einführung der Umweltzone erklärt werden. An dieser Messstation beträgt der Anteil des lokalen Kfz-Verkehrs an der $PM_{10}$-Feinstaubbelastung lediglich 6% und ist damit deutlich niedriger als am Stachus (29%) oder an der Landshuter Allee (45%). Die Heterogenität der Effektschätzer sowie die fehlende statistische Signifikanz könnten durch die relativ kurzen Beobachtungsperioden von jeweils nur 4 Monaten verursacht sein.

In einer weiteren Analyse haben Morfeld und Kollegen (2014) die $PM_{10}$-Konzentrationen von Messstationen innerhalb und außerhalb der Umweltzonen in 19 deutschen Städten analysiert. Alle untersuchten Umweltzonen waren aktiv in der Stufe 1 (gesperrt für Fahrzeuge der Schadstoffgruppe 1, ohne Plakette). Leider wurde in dieser Studie auf Einzeldarstellungen zugunsten der Gesamtauswertung verzichtet. Dabei ist zu berücksichtigen, dass die Gesamtauswertung dieser Studie sehr unterschiedliche Umweltzonen umfasst. Manche wurden in Großstädten wie Berlin, München oder Köln eingerichtet, andere in Kleinstädten (Herrenberg, Ilsfeld). Auch hinsichtlich der Größe unterschieden sich die untersuchten Umweltzonen gravierend. Während sie in Berlin, München oder Stuttgart eher groß sind, sind sie in anderen Städten verhältnismäßig klein (Augsburg, Köln, Ruhrgebietsstädte vor dem 01.01.2012). Eine Gesamtanalyse gibt zwar den mittleren Schätzer wider, kann aber die „Umweltzone“ als Maßnahme wegen die Heterogenität der untersuchten Umweltzonen nicht richtig bewerten. Eine detaillierte Darstellung der Ergebnisse (entweder für jede Umweltzone separat oder für unterschiedliche Umweltzonenklassen) würde eine richtige Bewertung dieser Maßnahme erlauben, da die Minderungseffekte sicher nicht für alle Umweltzonen gleich sind. Diese getrennte Darstellung der Minderungspotentiale fehlt in der Analyse von Morfeld und Kollegen (2014).

Morfeld und Kollegen (2014) haben als besten Effektschätzer (an allen Indexstationen) eine $PM_{10}$-Feinstaubreduktion von $\leqslant$0,2 µg/m³ (bzw. relative $PM_{10}$-Reduktionen $\leqslant$1%) ermittelt. Der beste Effektschätzer an allen Verkehrsstationen (also ohne städtische Hintergrund- und Industriestationen) lag unterhalb von 1 µg/m³ (bzw. weniger als 5%). Damit liegen die Ergebnisse durchaus in dem Bereich, der in vielen Wirkungsanalysen für die erste Stufe der Umweltzonen prognostiziert wurde (nicht höher als 1% im Innenstadtbereich und nicht höher als 5% an verkehrsbelasteten Stationen).

## 2.2 Beschreibung der Maßnahmen in Augsburg, München und Berlin

Im Rahmen dieses Vorhabens wurde ein Regressionsmodell für die Untersuchung der Effektivität der Umweltzonen entwickelt und auf drei Umweltzonen angewendet, die in Berlin, München und Augsburg eingeführt worden sind. Die drei Umweltzonen in Berlin, München und Augsburg unterscheiden sich hinsichtlich der Schärfe der Regelungen sowie ihrer Größe und wurden zu unterschiedlichen Zeitpunkten eingeführt (Tabelle 2). Während die Umweltzonen in Berlin und München relativ groß sind (88 und 44 km²) ist die Umweltzone in Augsburg eher klein (6 km²). Zusätzlich zu der Einführung der Umweltzone in München und Augsburg wurde in diesen Städten ein „Lkw-Durchfahrtsverbot“ erlassen, jedoch nicht in Berlin. In München ist das gesamte Stadtgebiet für die Durchfahrt von Kraftfahrzeugen über 3,5 Tonnen seit dem 1. Februar 2008 verboten, insbesondere wurde der Mittlere Ring für diese Fahrzeuge gesperrt. Die Ableitung des Lkw-Durchgangsverkehrs, d.h. aller Lkw, die nur Durchfahrtsverkehr darstellen und kein Ziel im Stadtgebiet haben, erfolgt soweit wie möglich auf kürzestem Wege zurück auf die Autobahnen und wird um das Münchner Stadtgebiet geleitet. Der relativ kurze Zeitabschnitt, in der nur das „Lkw-Durchfahrtsverbot“ als Maßnahme wirksam war, wurde von der Analyse ausgeschlossen. In Augsburg wurde ein „Lkw-Durchfahrtsverbot“ gleichzeitig mit der Einführung der Umweltzone am 1. Juli 2009 eingeführt und betrifft nur das Gebiet der Umweltzone.

Tabelle 1: Vergleich der Umweltzonen in Augsburg, München und Berlin.

| | Augsburg | München | Berlin |
|---|---|---|---|
| Fläche (% der Gesamtfläche) | ~ 6 km² (3%) | ~ 44 km² (14%) | ~ 88 km² (10%) |
| Einwohner innerhalb der Umweltzone (% der Gesamtpopulation) | ~ 20.000 (7%) | ~ 420.000 (32%) | ~ 1.000.000 (29%) |
| Einführung von Lkw-Durchfahrtsverbot | 01.07.2009 | 01.02.2008 | nicht geplant |
| Beginn der Stufe 1 | 01.07.2009 | 01.10.2008 | 01.01.2008 |
| Beginn der Stufe 2 | 01.01.2011 | 01.10.2010 | - |
| Beginn der Stufe 3 | 01.06.2016 | 01.10.2012 | 01.10.2010 |

## 2.3 Beurteilung der Wirksamkeit von Umweltzonen in Augsburg, München und Berlin anhand Immissionsmessdaten von $PM_{10}$-Feinstaub unter Verwendung eines Regressionsmodells

Als Modell für die statistische Analyse wurde ein semiparametrisches Modell mit autoregressiven Störtermen erster Ordnung benutzt:

$$\log(PM_{10}X) = \beta_0 + \beta_1 \log(PM_{10}Ref) + \beta_{W0} \cdot I_{W0} + \beta_{S1} \cdot I_{S1} + \beta_{W1} \cdot I_{W1} + \beta_{S2} \cdot I_{S2} + \beta_{W2} \cdot I_{W2} + f_{S0} \cdot I_{S0}(\text{Stunde}) + f_{W0} \cdot I_{W0}(\text{Stunde}) + f_{S1} \cdot I_{S1}(\text{Stunde}) + f_{W1} \cdot I_{W1}(\text{Stunde}) + f_{S2} \cdot I_{S2}(\text{Stunde}) + f_{W2} \cdot I_{W2}(\text{Stunde}) + f_{WD}(\text{Windrichtung}) + \beta_2(\text{Feiertag}) + \varepsilon$$

Als Zielvariable des Modells wurde die $PM_{10}$-Konzentration an einer Verkehrsstation verwendet ($PM_{10}X$). $PM_{10}Ref$ steht für die $PM_{10}$-Messwerte an der Referenzstation. Um jahreszeitliche Schwankungen in das Modell miteinzubeziehen, wurde die Auswirkung der Einführung der verschiedenen Stufen der Umweltzone (0, 1, 2) getrennt für Sommer (S) und Winter (W) analysiert. Als Referenzkategorie wurde “Sommer und keine Umweltzone” gewählt. $I_{S0}$, $I_{S1}$, $I_{S2}$ bezeichnen jeweils die Indikatorfunktionen für die Zeitperioden “Sommer und keine Umweltzone”, “Sommer und Umweltzone Stufe 1” und “Sommer und Umweltzone Stufe 2”. Entsprechend bezeichnen die Indikatorfunktionen $I_{W0}$, $I_{W1}$, $I_{W2}$: “Winter und keine Umweltzone”, “Winter und Umweltzone Stufe 1” und “Winter und Umweltzone Stufe 2”. Spezifische Schwankungen des Tages und der Tageszeit wurden durch einen wochenstündlichen Trend (Stunde) modelliert. Zu diesem Zweck wurden als Basisfunktionen zyklische penalisierte Splines benutzt, um eine glatte Funktionsschätzung zu erhalten. Für die Schätzung des Effekts der Windrichtung an den einzelnen Messstationen wurden ebenfalls zyklische penalisierte Splines verwendet. Da angenommen werden kann, dass sich an

Feiertagen das Verkehrsaufkommen ändert, wurde ein Indikator für Feiertage an Werktagen mit in das Modell aufgenommen.

In Augsburg standen Daten von vier Messstationen zur Verfügung: Haunstetten (Referenzmessstation), Königsplatz und Karlstrasse (Verkehrsmessstationen) und Bourgesplatz (städtischer Hintergrund). In München wurde die Messstation in Johanniskirchen als Referenzmessstation benutzt, Landshuter Allee und Stachus repräsentieren Verkehrsmessstationen und Lothstrasse ist eine Messstation im städtischen Hintergrund. In Berlin wurde die Messstation in der Nansenstrasse als Referenzmessstation in der Analyse verwendet, Frankfurter Allee und Schildhornstrasse werden als Verkehrsmessstationen klassifiziert.

Die Ergebnisse der Modellierungen sind in den Tabelle 2 bis Tabelle 4 dargestellt.

Tabelle 2: Änderungen der $PM_{10}$-Konzentrationen[a] in Augsburg nach der Einführung der Umweltzone in Stufe 1 und 2 (verglichen mit der Referenzperiode vor der Einführung der Umweltzone). Fläche der UWZ: 6 km².

| Standort | Sommer | | | Winter | | |
|---|---|---|---|---|---|---|
| **Umweltzone Stufe 1** | | | | | | |
| | Effekt | 95% Konfidenzinterval | p-Wert | Effekt | 95% Konfidenzinterval | p-Wert |
| Königsplatz (sv) | **-11.1%** | **(-20.0%, -1.4%)** | **<0.001** | **-10.8%** | **(-19.6%, -0.9%)** | **<0.001** |
| Karlstrasse (sv) | **-6.6%** | **(-12.2%, -0.7%)** | **0.03** | 3.0% | (-3.2%, 9.7%) | <0.05 |
| Bourgesplatz (sH) | -1.2% | (-12.6%, 11.6%) | 0.84 | 5.5% | (-6.0%, 18.5%) | <0.005 |
| **Umweltzone Stufe 2** | | | | | | |
| Königsplatz (sv) | **-12.4%** | **(-21.1%, -2.7%)** | **<0.001** | **-7.0%** | **(-16.2%, 3.3%)** | **<0.001** |
| Karlstrasse (sv) | **-7.5%** | **(-13.2%, -1.4%)** | **0.02** | **9.2%** | **(2.5%, 16.2%)** | **<0.001** |
| Bourgesplatz (sH) | 2.0% | (-7.9%, 13.1%) | 0.70 | **18.8%** | **(7.0%, 31.8%)** | **<0.001** |

[a] adjustiert für $PM_{10}$-Konzentrationen an der Referenzmessstation, Windrichtung, Wochentag, Wochenstunde und Feiertage. sv: städtisch verkehrsnah, sH: städtischer Hintergrund.

Tabelle 3: Änderungen der $PM_{10}$-Konzentrationen[a] in München nach der Einführung der Umweltzone in Stufe 1 und 2 sowie des Lkw-Durchfahrtsverbots (verglichen mit der Referenzperiode vor der Einführung der Umweltzone). Fläche der UWZ: 44 km².

| Standort | Sommer | | | Winter | | |
|---|---|---|---|---|---|---|
| **Umweltzone Stufe 1 und Lkw-Durchfahrtsverbot** | | | | | | |
| | Effekt | 95% Konfidenzinterval | p-Wert | Effekt | 95% Konfidenzinterval | p-Wert |
| Landshuter Allee (sv) | **-6.4%** | **(-11.2%, -1.6%)** | **0.01** | -3.7% | (-8.5%, 1.2%) | 0.15 |
| Stachus (sv) | 2.4% | (-0.8%, 5.6%) | 0.11 | -1.3% | (-4.4%, 1.8%) | 0.41 |
| Lothstr. (sH) | **-6.4%** | **(-10.2%, -2.6%)** | **0.00** | **-5.0%** | **(-8.8%, -1.2%)** | **0.01** |
| **Umweltzone Stufe 2 und Lkw-Durchfahrtsverbot** | | | | | | |
| Landshuter Allee (sv) | **-14.9%** | **(-19.2%, -10.5%)** | **‹0.001** | **-5.9%** | **(-10.7%, -1.1%)** | **0.02** |
| Stachus (sv) | **-6.4%** | **(-9.4%, -3.5%)** | **‹0.001** | **-3.0%** | **(-6.0%, 0.1%)** | **0.06** |
| Lothstr. (sH) | **-10.4%** | **(-14.1%, -6.7%)** | **‹0.001** | **-6.3%** | **(-10.1%, -2.5%)** | **0.00** |

[a] adjustiert für $PM_{10}$-Konzentrationen an der Referenzmessstation, Windrichtung, Wochentag, Wochenstunde und Feiertage. sv: städtisch verkehrsnah, sH: städtischer Hintergrund.

Tabelle 4: Änderungen der $PM_{10}$-Konzentrationen[a] in Berlin nach der Einführung der Umweltzone in Stufe 1 und 3 (verglichen mit der Referenzperiode vor der Einführung der Umweltzone). Fläche der UWZ: 88 km². (Ungar, 2014).

| Standort | Sommer | | | Winter | | |
|---|---|---|---|---|---|---|
| **Umweltzone Stufe 1** | | | | | | |
| | Effekt | 95% Konfidenzinterval | p-Wert | Effekt | 95% Konfidenzinterval | p-Wert |
| Schildhornstr. (sv) | **-6.9%** | **(-10.4%, -3.4%)** | **‹0.001** | **-12.7%** | **(-16.0%, -9.4%)** | **‹0.001** |
| Frankfurter Allee (sv) | **-8.1%** | **(-10.8%, -5.4%)** | **‹0.001** | **-5.8%** | **(-8.5%, -3.0%)** | **‹0.001** |
| **Umweltzone Stufe 3** | | | | | | |
| Schildhornstr. (sv) | **-19.0%** | **(-21.7%, -16.3%)** | **‹0.001** | **-13.3%** | **(-16.2%, -10.4%)** | **‹0.001** |
| Frankfurter Allee (sv) | **-16.0%** | **(-18.2%, -13.9%)** | **‹0.001** | **-9.1%** | **(-11.4%, -6.7%)** | **‹0.001** |

[a] adjustiert für $PM_{10}$-Konzentrationen an der Referenzmessstation, Windrichtung, Wochentag, Wochenstunde und Feiertage. sv: städtisch verkehrsnah, sH: städtischer Hintergrund.

## 2.4 Beurteilung der Wirksamkeit von Umweltzonen in Deutschland anhand Immissionsmessdaten von Gesamtkohlenstoff unter Verwendung eines Regressionsmodells

Die in der Öffentlichkeit geführte Diskussion über die Wirksamkeit der Umweltzone als Maßnahme für eine verbesserte Luftqualität ist oft nur vom Blick auf die gemessenen $PM_{10}$-Konzentrationen geprägt. Ein Grundproblem des Wirksamkeitsnachweises durch $PM_{10}$-Immissionsdaten besteht darin, dass $PM_{10}$ (und in geringerem Umfang auch $PM_{2,5}$) Massenkonzentrationen lokal nur relativ wenig durch Abgasemissionen aus dem Kfz-Verkehr beeinflusst werden. Der Hauptanteil des $PM_{10}$-Feinstaubes wird im Allgemeinen durch relativ hohe Beiträge der atmosphärisch langlebigen Partikel im Größenbereich 0,2 bis 1 µm (in dem sogenannten Akkumulationsmode) gebildet. Einen wichtigen Bestandteil des Feinstaubes bilden größere Partikel (> 1 µm), die durch Aufwirbelung von Bodenstaub sowie durch Abrieb von Reifen, Bremsen und Fahrbahn entstehen. Abgasemissionen enthalten vorwiegend Rußpartikel, die meist kleiner als 0,2 µm sind und in sehr großer Anzahl emittiert werden. An der großen Oberfläche dieser Rußpartikel lagern sich toxische Produkte des Verbrennungsprozesses an. Diesem kleinen Feinstaubanteil wird eine erhebliche Gesundheitsrelevanz zugeschrieben. Es ist davon auszugehen, dass die Einführung von Umweltzonen insbesondere den hochtoxischen Dieselrußanteil im Feinstaub reduziert. Durch die Messung von Ruß-Konzentrationen in der Außenluft könnte man somit die Wirksamkeit der Umweltzonen viel genauer untersuchen, als durch Messungen von $PM_{10}$-Feinstaubkonzentrationen.

Leider werden derzeit in Deutschland die Ruß-Konzentrationen in der Außenluft routinemäßig nicht gemessen, da es bisher keine Grenzwerte für Rußpartikel gibt. Außerdem gibt es kein Standard-Messverfahren für die Ruß-Messungen. Die bisher verwendeten unterschiedlichen Methoden der Ruß-Bestimmung liefern zum Teil unterschiedliche Ergebnisse. Eine Ausnahme stellt Berlin dar, wo seit den neunziger Jahren die Konzentration von kohlenstoffhaltigen Partikeln an Hauptverkehrsstraßen gemessen wird. So sind für Berlin Konzentrationen von elementarem Kohlenstoff (EC) und organischem Kohlenstoff (OC) erhältlich. Somit konnten die Effekte der Einführung der Umweltzone für Berlin auch für diese Parameter unter Verwendung eines angepassten Regressionsmodells untersucht werden. Das Modell musste angepasst werden, da in dem Datensatz keine stündlichen Mittelwerte sondern Mittelwerte für 14 Tage (2 Wochen) vorliegen. Somit können die spezifischen Schwankungen des Tages und der Tageszeit nicht berücksichtigt werden und die entsprechenden Terme in der Modell-Gleichung fallen weg. Dafür wurde in diesem Modell auf zusätzliche meteorologische Parameter wie Globalstrahlung, Windgeschwindigkeit und Temperatur adjustiert. Als Zielvariable wurde die Konzentration vom Gesamtkohlenstoff (TC, Total carbon) verwendet. Der Gesamtkohlenstoffgehalt (TC) in der Atmosphäre ergibt sich aus der Addition von organischer Masse (OM) und EC. Die Konzentration der organische Masse wird durch Multiplikation der Konzentration von organischen Kohlenstoff OC mit Faktor 1,2 errechnet (OM=1,2 x OC). Da bei der Messung der Kohlenstoffarten (OC, EC) Unsicherheiten in der Unterscheidung der einzelnen Komponenten auftreten, ist der Gesamtkohlenstoffgehalt (TC) der mit der geringsten Unsicherheit versehene Parameter. Die endgültige Modell-Gleichung sieht folgendermaßen aus:

$$\log(TC_X) = \beta_0 + \beta_1 \log(TC_{ref}) + \beta_{UZ1} \cdot I_{UZ1} + \beta_{UZ3} \cdot I_{UZ3} + \beta_{S0} \cdot I_{S0} + \beta_{W0} \cdot I_{W0} + \beta_{S1} \cdot I_{S1} + \beta_{W1} \cdot I_{W1} + \beta_{S3} \cdot I_{S3} + \beta_{W3} \cdot I_{W3} + f_{WD}(\text{Windrichtung}) + f_{WS}(\text{Windgeschwindigkeit}) + f_{DR}(\text{Globalstrahlung}) + f_T(\text{Temperatur}) + \beta_2(\text{Feiertag}) + \varepsilon$$

Die Ergebnisse der Modellierungen sind in Tabelle 5 dargestellt.

Tabelle 5: Änderungen der TC-Konzentrationen[a] in Berlin nach der Einführung der Umweltzone in Stufe 1 und 3 (verglichen mit der Referenzperiode vor der Einführung der Umweltzone).

| Standort | Sommer | | | Winter | | |
|---|---|---|---|---|---|---|
| **Umweltzone Stufe 1** | | | | | | |
| | Effekt | 95% Konfidenzintervall | p-Wert | Effekt | 95% Konfidenzintervall | p-Wert |
| Schildhornstr. (sv) | **-13.5%** | **(-23.0%, -4.0%)** | **0.01** | -8.7% | (-19.4%, 2.0%) | 0.13 |
| Frankfurter Allee (sv) | **-11.4%** | **(-21.8%, -0.9%)** | 0.05 | **-12.3%** | **(-23.3%, -1.3%)** | **0.04** |
| **Umweltzone Stufe 3** | | | | | | |
| Schildhornstr. (sv) | **-25.1%** | **(-31.1%, -19.0%)** | **‹0.001** | **-15.8%** | **(-23.3%, -8.4%)** | **‹0.001** |
| Frankfurter Allee (sv) | **-23.0%** | **(-29.8%, -16.2%)** | **‹0.001** | **-18.8%** | **(-26.4%, -11.2%)** | **‹0.001** |

[a] adjustiert für TC-Konzentrationen an der Referenzmessstation, Jahreszeiten und Windrichtung.
sv: städtisch verkehrsnah.

Im Vergleich zum Rückgang der $PM_{10}$-Konzentration ist die Reduktion von TC-Konzentration deutlicher (s. Tabelle 4). So sinkt die $PM_{10}$-Belastung im Winter nach der Einführung der Stufe 3 der Umweltzone um 9% in der Frankfurter Allee und um 13% in der Schildhornstrasse. Die Reduktion von TC erreicht 16% in der Frankfurter Allee und fast 17% in der Schildhornstrasse. Auch die entsprechenden Reduktionen im Sommerhalbjahr sind deutlicher ausgeprägt für TC als für $PM_{10}$. Bei diesem Vergleich muss man allerdings beachten, dass aufgrund der unterschiedlichen Datenstruktur die Ergebnisse der Modellierung nicht direkt vergleichbar sind.

## 2.5 Diskussion der Ergebnisse

### 2.5.1 Vergleich der drei Umweltzonen Augsburg, München und Berlin

Die Ergebnisse der Modellierung für Augsburg ergeben kein einheitliches Bild. Eine statistisch signifikante Minderung der $PM_{10}$-Belastung wurde nur an der Verkehrsmessstation am Königsplatz beobachtet, während die Minderung in der Karlstraße (Straßenschlucht) nur im Sommer sichtbar wird (im Winter gibt es sogar einen statistisch signifikanten Anstieg der $PM_{10}$-Konzentrationen). Keine Änderungen der $PM_{10}$-Konzentrationen sind an der Hintergrundmessstation im Sommer zu beobachten, während im Winter wie schon in der Karlstrasse eher ein Anstieg der $PM_{10}$-Belastung zu verzeichnen ist. Darüber hinaus ist kein Unterschied zwischen der Umweltzone in Stufe 1 und Stufe 2 bezüglich der Änderungen der $PM_{10}$-Konzentratioen sichtbar.

In München sind die Minderungseffekte erst nach der Verschärfung der Regelungen von Stufe 1 auf Stufe 2 (gelbe und grüne Plaketten zugelassen, rote ausgesperrt) an allen Messstationen statistisch signifikant. Für die erste Stufe der Umweltzone sieht man keine Minderungseffekte am Stachus. Allerdings wurden in der Nähe der Messstation am Stachus im Beobachtungszeitraum „Umweltzone Stufe 1“ umfangreife Sanierungsarbeiten des S-Bahn und U-Bahn Bahnhofs sowie Erneuerungen der Straßenbahngleise durchgeführt, so dass diese möglicherweise zu einer Zusatzbelastung an dieser Messstation geführt haben konnten.

Die Minderungseffekte der Umweltzone in München sind deutlich stärker nach der Verschärfung der UWZ-Regelungen von Stufe 1 auf Stufe 2. Insgesamt ist die Abnahme der $PM_{10}$-Belastung im Sommer größer (bis zu 15% an der Landshuter Allee) als im Winter.

Auch in Berlin sind die Minderungseffekte nach der Verschärfung der Regularien (Stufe 3) deutlicher als in der ersten Stufe. Dieser Anstieg ist besonders im Sommer sichtbar. So hat sich die Abnahme der $PM_{10}$-Konzentrationen nach der Einführung der Stufe 3 in Berlin an der Messstation Frankfurter Allee verdoppelt (von -8% für Stufe 1 auf -16% für Stufe 3). Dieser Trend ist schwächer ausgeprägt im Winter.

### 2.5.2 Vergleich Sommer – Winter

Insgesamt sind die Effekte der Umweltzone auf die Veränderung der $PM_{10}$-Konzentrationen im Sommerhalbjahr immer deutlicher und stärker als im Winterhalbjahr. Möglicherweise wird der Minderungseffekt der Umweltzone im Winter durch andere, zusätzliche Partikelquellen (wie zum Beispiel Hausbrand, Holzverbrennung, Aufwirbelung von trockenem Streusalz) im Winter überdeckt. Somit steigt der Anteil der „Grobfraktion" (auch als „coarse particulate fraction" bezeichnet) im $PM_{10}$-Feinstaub an, die sich im Größenbereich von 2,5 bis 10 Mikrometer befindet. Die Partikel aus dem Auspuff, deren Ausstoß durch die Umweltzone reduziert wird, befinden sich hauptsächlich in der $PM_{2,5}$-Fraktion des Feinstaubes. Der Anteil dieser Fraktion am $PM_{10}$-Feinstaub ist geringer im Winter als im Sommer. Darüber hinaus kommt es im Winter vermehrt zu Inversionswetterlagen und dadurch zu einer Ansammlung von Luftschadstoffen in der unteren Atmosphärenschicht. Offensichtlich ist der reduzierende Einfluss von Maßnahmen wie Umweltzonen während solcher Episoden stark eingeschränkt.

### 2.5.3 Vergleich $PM_{10}$ – Gesamtkohlenstoff (total carbon, TC) und andere Parameter

Wie aus den Ergebnissen in Tabelle 5 sichtbar, ist die Abnahme der Gesamtkohlenstoff-Konzentration deutlich stärker ausgefallen als die $PM_{10}$-Minderung. Ähnliche Ergebnisse wurden auch von anderen deutschen Städten berichtet. Es wurden vor einiger Zeit Studien aus Leipzig und München publiziert, die über Ergebnisse von Sondermesskampagnen berichten, in denen unter anderem auch die Konzentration von Dieselruß bestimmt wurde.

In der Studie von Qadir und Kollegen (2013) wurde in München $PM_{2,5}$-Feinstaub im Winter 2006/07 (vor der Einführung der Umweltzone) und Winter 2009/2010 (nach der Einführung der Umweltzone) auf Filtern gesammelt und dann auf organische Komponenten und EC/OC-Gehalt analysiert. Um die wichtigsten Partikelquellen zu identifizieren wurde die Methode der Positiven Matrix Faktorisierung (PMF) angewendet. Es wurden unterschiedliche Faktoren als Partikelquellen identifiziert und quantifiziert (wie zum Beispiel „Kfz-Verkehr", „Verbrennung fester Fossilstoffe", „Kochen" und andere). Der Beitrag des Faktors „Kfz-Verkehr" zu der Gesamtkonzentration von $PM_{2,5}$-Feinstaub sank nach der Einführung der Umweltzone in München um 60%. Außerdem nahm die mittlere EC-Konzentration in dem Faktor „Kfz-Verkehr" von 1,1 (vor der Einführung der Umweltzone) auf 0,5 $\mu g/m^3$ ab (nach der Einführung der Umweltzone), was eine Abnahme um 50% bedeutet.

In Leipzig wurde die Einführung der Umweltzone messtechnisch begleitet. In Sondermesskampagnen wurde zusätzlich zu $PM_{10}$ und $PM_{2,5}$ auch die Bestimmung von Dieselabgasen (Ruß gemessen sowohl als EC als auch als BC) durchgeführt. Die Sondermessungen haben an den meisten verkehrsnahen Messstationen einen deutlichen Rückgang der Konzentrationen von EC und BC nachgewiesen. An einer Messstation im Zentrum der Umweltzone war die relative Minderung von EC um 24 % und von BC um 31 % nachweisbar (Löschau, 2013).

Damit zeigt sich deutlich, dass die Beurteilung der Wirksamkeit von Umweltzonen anhand von Messungen der Konzentrationsänderungen von $PM_{10}$-Feinstaub in der Außenluft nur bedingt möglich ist. Die Umweltzonen wurden zwar primär als Maßnahme für die Reduktion der $PM_{10}$-Belastung eingeführt (weil diese gesetzlich reguliert wird), haben aber deutlicheren Einfluss auf die Reduktion des toxisch relevanten Rußanteils. Die Überwachung des toxisch relevanten Anteils des $PM_{10}$-Feinstaubs wäre eine bessere Vorgehensweise, um die Wirksamkeit der Umweltzonen zu beurteilen. Zusätzlich wäre es sehr wichtig, die Veränderung der Konzentrationen anderer Partikelgrößenklassen, wie $PM_{2,5}$-Feinstaubs oder ultrafeine Partikel, zu bestimmen. Die Reduktion dieser Partikelparameter aufgrund der Einführung von Umweltzonen sollte ebenfalls deutlicher ausfallen als für $PM_{10}$. Außerdem haben diese Partikelfraktionen möglicherweise stärkere gesundheitliche Effekte auf die Bevölkerung, die über die Wirkungen von $PM_{10}$ hinausgehen.

# 3 Luftqualitätsmodellierung in Augsburg

In epidemiologischen Langzeitstudien, in denen die Zusammenhänge zwischen der Langzeitexposition zu Luftschadstoffen und gesundheitlichen Effekten untersucht werden, ist die Abschätzung der räumlichen Verteilung der Luftschadstoffe von großer Bedeutung (Anderson 2009). Die Anzahl der fest installierten Messstationen in einem Studiengebiet ist in der Regel zu klein, um die räumliche Verteilung der Schadstoffe messtechnisch abzubilden. Somit ist der Einsatz von Modellen zur Bestimmung der räumlich-zeitlichen Variation der Luftschadstoffe unverzichtbar. Der große Vorteil der Modellierung, ungeachtet der Limitierung durch technische, ökonomische oder zeitliche Rahmenbedingungen, ist die Möglichkeit, die Konzentrationen der Luftschadstoffe in einer hohen räumlichen Auflösung zu ermitteln. Sehr oft wird die Berechnung der Konzentration an ausgewählten Punkten mit sogenannten Ausbreitungsmodellen durchgeführt.

In diesem Vorhaben wurden $PM_{2,5}$ und $NO_x$ (Stickoxide) Konzentrationen mit Hilfe von zwei unterschiedlichen Modellen, IMMIS$^{luft}$ und AirViro, simuliert. IMMIS$^{luft}$ ist ein Screening-Programm zur Bestimmung der Luftschadstoff-Emissionen und -Immissionen in Innenstädten (http://www.immis.de/). IMMIS$^{luft}$ berechnet die durch Kraftfahrzeuge erzeugten Emissionen und modelliert die Ausbreitung der Immissionen von Luftschadstoffen im Straßenraum. Es beruht auf dem CPB-Modell (Canyon Plume-Box) für Straßenschluchten und einem Box-Modell für offene Bebauungen. Als Modell-Input werden üblicherweise eine 10 Jahres-Klimatologie des Deutschen Wetterdienstes (DWD), Daten zur Straßengeometrie sowie straßenspezifische Daten zur Verkehrszusammensetzung und -stärke verwendet. Die IMMIS Modelle sind in die GIS-Software ArcGIS integriert (Geographisches Informationssystem GIS). Somit ist die Berechnung von Kraftfahrzeug-Emissionen und Immissionen von Luftschadstoffen im Straßenraum zur anschließenden Visualisierung, Analyse, kartographischen Aufbereitung und Karten-Publikation direkt im ArcGIS möglich.

AirViro ist ein Gaußmodell zur Bestimmung der verkehrsbedingten Belastung durch Straßennetze unter Nutzung von Immissions- und Emissionsdaten, sowie Dispersions- und Windmodellierung (http://www.airviro.smhi.se).

Die Ergebnisse der Modellierung wurden mit den Messdaten verglichen und validiert.

## 3.1 IMMIS-Modell

In Tabelle 6 sind die Ergebnisse der IMMIS Modellierung für drei Standorte in der Augsburger Innenstadt dargestellt. Die Karlstraße ist eine Hauptverkehrsstraße die dem Charakter einer Straßenschlucht ähnelt. In dieser Straße befindet sich eine Messstation des Lufthygienischen Landesüberwachungssystems Bayern (LÜB), das vom Bayerischen Landesamt für Umwelt betrieben wird. Die Karolinenstraße ist eine Nebenstraße mit deutlichen wenig Kfz-Verkehr, aber einem hohen Anteil an LKW-Verkehr, der überwiegend gewerblich bedingt ist. Obstermarkt ist eine Nebenstraße mit wenig LKW-Verkehr.

Die drei Standorte sind nicht weit voneinander entfernet, trotzdem sind die ermittelten verkehrsbedingten Zusatzbelastungen sehr unterschiedlich. An der Karlstraße beträgt die durchschnittliche tägliche Verkehrsstärke (DTV) mehr als 26.000 Fahrzeuge/24h, die Zusatzbelastung an $PM_{10}$, $PM_{2,5}$ und $NO_x$ wurde auf 9,6, 6,3 und 72,3 µg/m$^3$ geschätzt.

Obwohl die deutlich kleinere DTV an den beiden Standorten in Nebenstraßen relativ vergleichbar ist (5.900 Kfz/24h in der Karolinenstraße, 4.900 Kfz/24h am Obstermarkt), unterscheiden sich die ermittelten verkehrsbedingten Zusatzbelastungen sehr deutlich, insbesondere für $NO_x$ (49.4 vs. 6.5 µg/m$^3$). Dies ist durch den höheren Anteil an LKW-Verkehr in der Karolinenstraße bedingt.

Tabelle 6: Modellierte verkehrsbedingte $PM_{10}$, $PM_{2,5}$ and $NO_x$ Zusatzbelastung an drei Standorten in der Augsburger Innenstadt.

| Straße | Karlstraße | Karolinenstraße | Obstermarkt |
|---|---|---|---|
| DTV* | 26,258 | 5,970 | 4,863 |
| % LKW Verkehr | 0.05 | 0.18 | 0.00 |
| Max. Geschwindigkeit (km/h) | 50 | 30 | 30 |
| $PM_{10}$ (µg/m³) | 9.80 | 4.60 | 0.90 |
| $PM_{2,5}$ (µg/m³) | 6.30 | 3.00 | 0.60 |
| NOx (µg/m³) | 72.30 | 49.40 | 6.50 |

*DTV = durchschnittliche tägliche Verkehrsstärke (in Kfz/24h)

Zum Vergleich sind in Tabelle 7 die $PM_{10}$, NO, $NO_2$ und $NO_x$ Konzentrationen dargestellt, die an den LÜB Messstationen am Standfort LfU (Bayerisches Landesamt für Umweltschutz) und in der Karlstraße gemessen wurden. Die Messstation LfU liegt ca. 4 km von der Stadtmitte entfernt im Süden der Stadt und wird als Hintergrund-Messstation im vorstädtischen Gebiet klassifiziert. Die Differenz zwischen den Werten der Hintergrundstation (LfU) und der Verkehrsmessstation in der Karlstraße beträgt 82 µg/m³ für $NO_x$ und 8 µg/m³ für $PM_{10}$. Diese Unterschiede sind hauptsächlich auf die Zusatzbelastung durch Kfz-Verkehr in der Karlstraße zurückzuführen und sind den modellierten Zusatzbelastungen sehr ähnlich (72 µg/m³ für $NO_x$ und 10 µg/m³ für $PM_{10}$, Tabelle 6).

Tabelle 7: Mittelwerte der NO, $NO_2$, $NO_x$ and $PM_{10}$-Konzentrationen (µg/m³) an zwei Messstationen für die Jahre 2008-2009.

| | NO | $NO_2$ | $NO_x$ | $PM_{10}$ |
|---|---|---|---|---|
| LfU (Hintergrund) | 7.70 | 21.10 | 28.80 | 20.80 |
| Karlstraße (Verkehrsexponiert) | 55.70 | 55.30 | 111.00 | 29.20 |
| **Differenz** | **48.00** | **34.20** | **82.20** | **8.40** |

## 3.2 AirViro Modell

Das AirViro-Modell liefert die räumliche Verteilung der Schadstoffe für das gesamte Stadtgebiet. Die Zusatzbelastungen sind spezifisch für einzelne Emittenten-Gruppen ermittelt. Um die Ergebnisse der Modellierung zu validieren, haben wir die modellierten Konzentrationen mit gemessenen Werten verglichen. Für den Vergleich wurde eine LÜB-Messstation am Bourgesplatz (BP) sowie eine Forschungsmessstation am Gelände der Hochschule Augsburg (HA) ausgewählt. Beide Standorte befinden sich im städtischen Hintergrund.

Abbildung 1 zeigt den Tagesgang der gemessenen und modellierten $NO_x$ Konzentration am der BP Messstation. Der Tagesgang der modellierten $NO_x$-Konzentrationen stimmt gut mit den gemessenen überein. Die gemessenen Konzentrationen sind selbstverständlich höher, da die Modelle den Beitrag der regionalen Quellen nicht berücksichtigen können.

Abbildung 1: Tagesgang der gemessenen $NO_x$ Gesamtkonzentration (grün) und modellierten lokalen Zusatzbelastung (blau) an der Messstation Bourgesplatz in Augsburg (Jahr 2008).

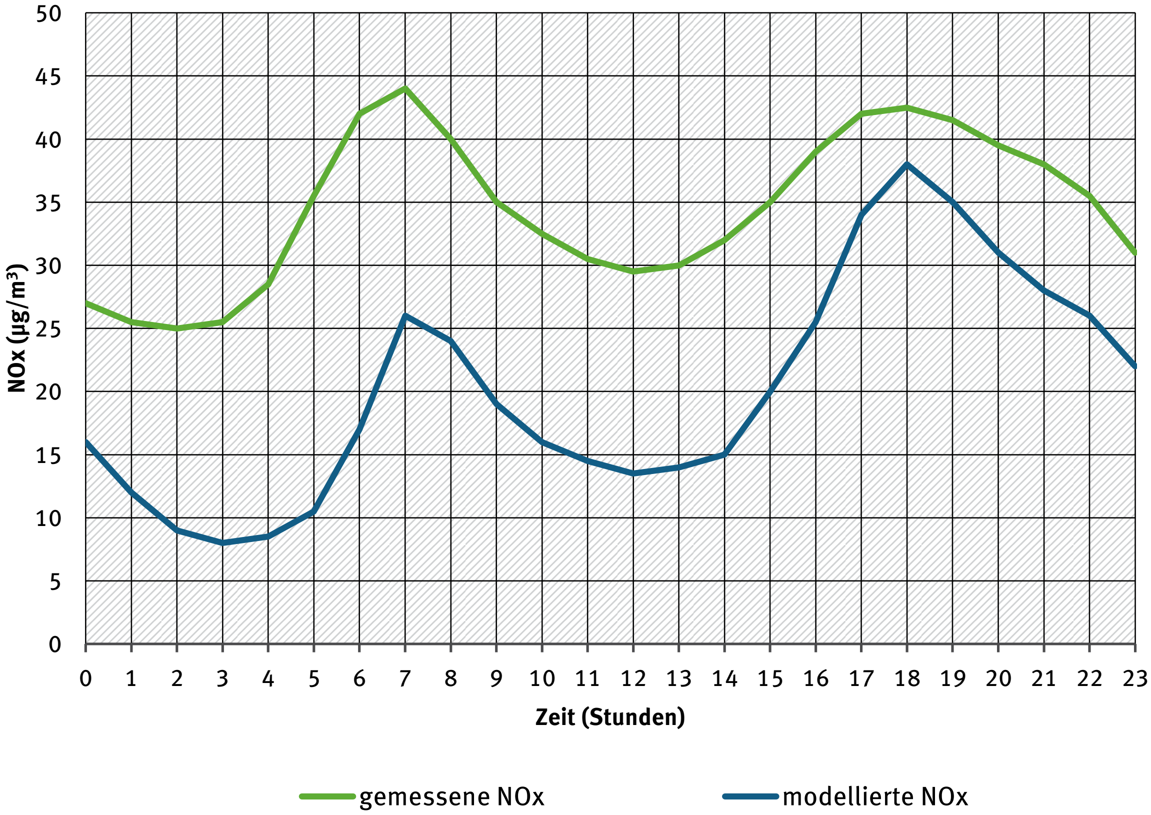
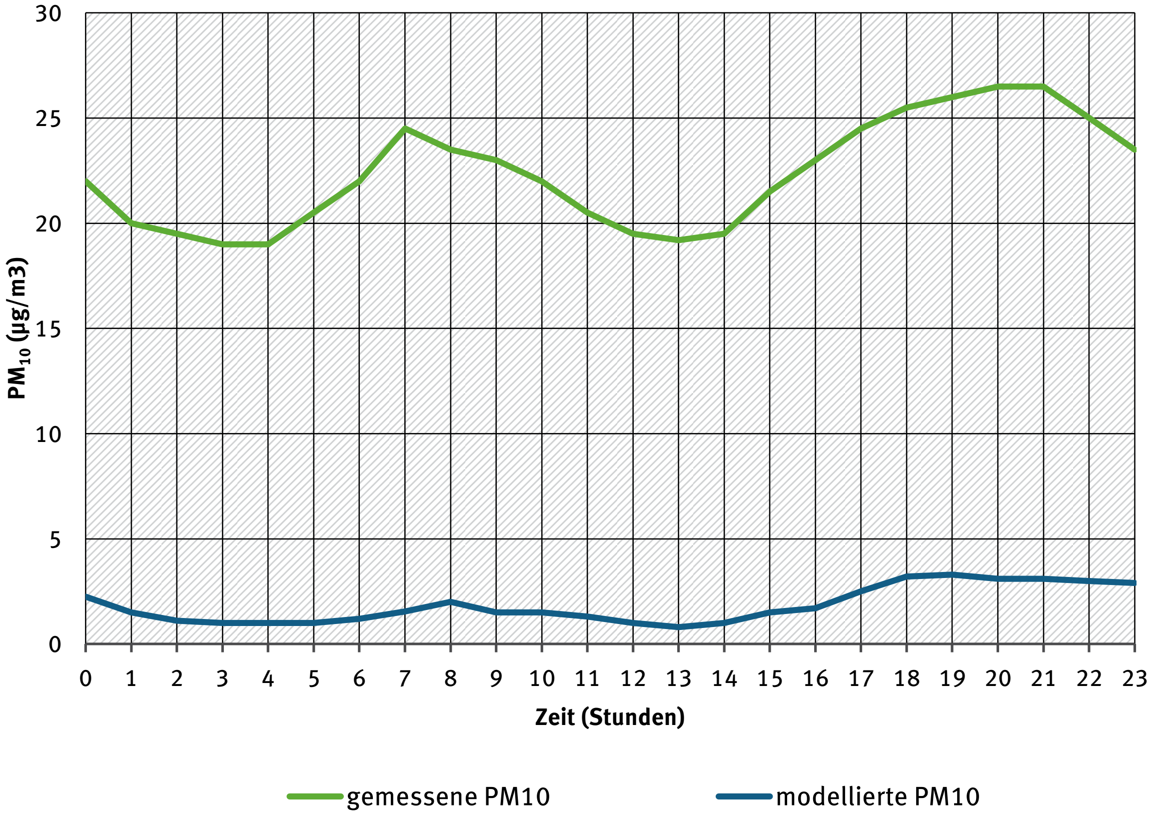



Abbildung 2 zeigt den Tagesgang der modellierten $PM_{10}$ Zusatzbelastung aus lokalen Quellen im Vergleich zu den gemessenen $PM_{10}$ Konzentration für die LÜB Messstation am Bourgesplatz (Bezugsjahr 2008).

Die gemessenen Konzentrationen sind viel höher als der Beitrag der lokalen $PM_{10}$-Quellen. So wurden im Jahre 2008 am BP 21,2 µg/m³ und am Standort HA 17,3 µg/m³ $PM_{10}$ gemessen. Die modellierte Zusatzbelastung aus lokalen Quellen betrug 1,8 µg/m³ am BP (9%) und 2,0 µg/m³ $PM_{10}$ (12%) am Standort HA. Bei diesen Angaben ist zu beachten, dass es sich um Standorte im städtischen Hintergrund handelt. Außerdem konnte bei der Modellierung nicht die Gesamtzusatzbelastung abgeschätzt werden, da Zuschläge für Abrieb und Aufwirbelung nicht in der Modellierung berücksichtigt wurden.

Abbildung 2: Tagesgang der gemessenen $PM_{10}$ Gesamtkonzentration (grün) und modellierten lokalen Zusatzbelastung (blau) an der Messstation Bourgesplatz in Augsburg (Jahr 2008).




## 3.3 Abschätzung der Wirksamkeit der Umweltzone in Augsburg durch das AirViro Modell

Die Wirkungsanalyse für die geplanten Maßnahmen der Umweltzone wurde für die drei Hintergrundstandorte BP, HA und LfU vorgenommen. Als Bezugsjahr wurde das Jahr 2008 (vor der Einführung der Umweltzone) ausgewählt. Es wurde angenommen, dass die erste Stufe der Umweltzone im Jahr 2009 (Kfz mit roter, gelber und grüner Plakette), die zweite Stufe im Jahr 2011 (Kfz mit gelber und grüner Plakette) und die dritte Stufe im Jahr 2013 (Kfz mit grüner Plakette) wirksam war. Die Emissionen aller anderen lokalen Schadstoffquellen wurden als konstant angenommen, verändert wurde nur der Anteil der Fahrzeuge, die gemäß der Umweltzonen-Regelung zugelassen waren. Die meteorologischen Bedingungen wurden bei der Berechnung der Immissionen ebenfalls als konstant angenommen. In Abbildung 3 sind die gerechneten $NO_x$ und $PM_{10}$ Emissionen dargestellt. In Abbildung 4 sind die sich daraus ergebenden Immissionswerte für $NO_x$ und $PM_{10}$ und für die drei untersuchten Standorte dargestellt. Der Rückgang der Luftschadstoffkonzentrationen beträgt bis 0.3 µg/m³ für $PM_{10}$ und 4 µg/m³ für $NO_x$. Die Reduktionen sind zwar zahlenmäßig klein, stellen jedoch immerhin etwa 16% der abgasbedingten $PM_{10}$-Zusatzbelastung am Bourgesplatz dar.

Abbildung 3: Änderungen der Gesamtemission für $NO_x$ (links) und $PM_{10}$ (rechts) durch die stufenweise Einführung der Umweltzone in Augsburg (Meteorologie des Jahres 2008 wurde für alle nachfolgenden Jahre als konstant angenommen, es wurden nur die Beiträge des lokalen KfZ-Verkehrs verändert).

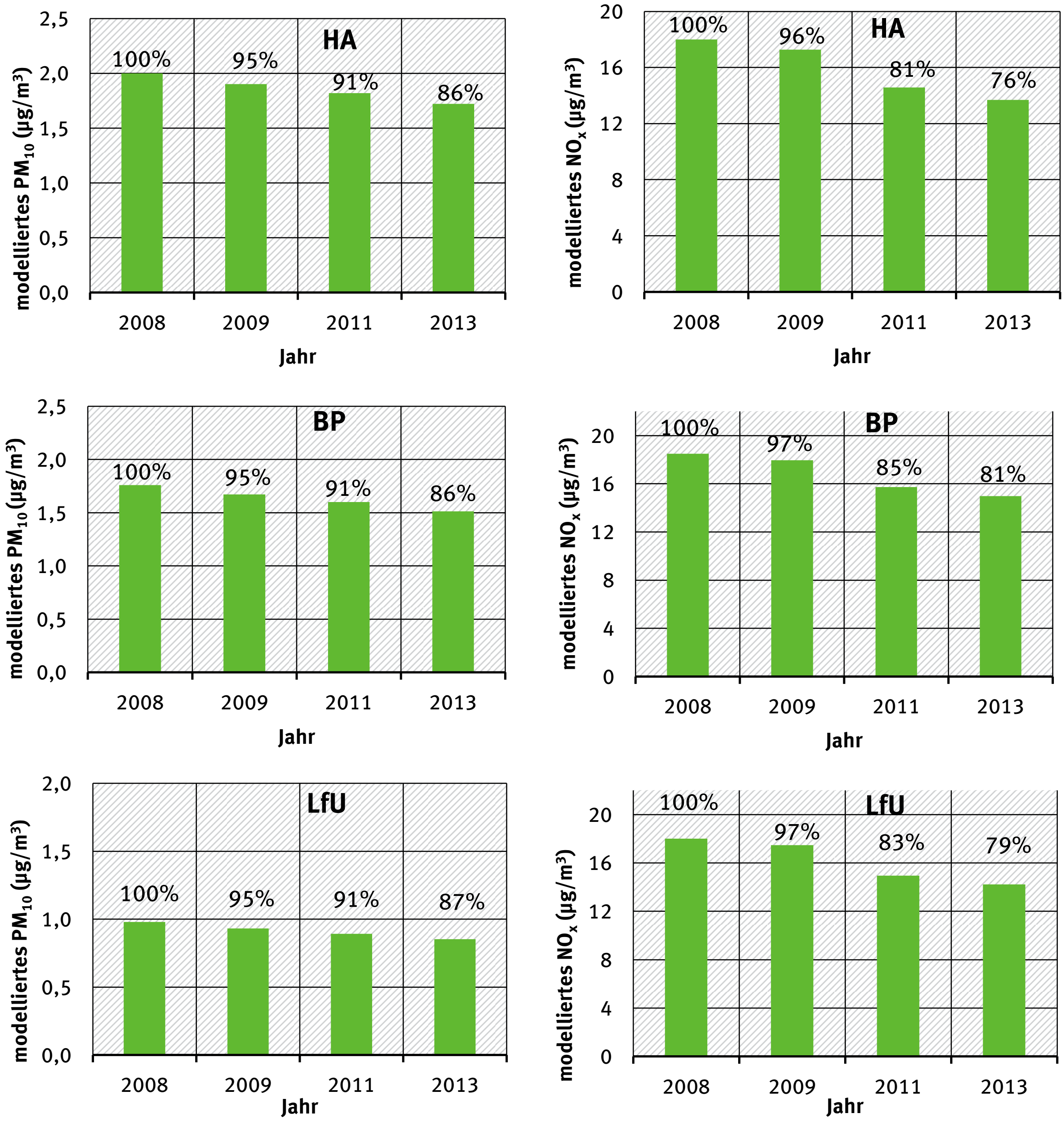

Auffallend ist die nahezu identische prozentuale Reduktion der Zusatzbelastung an allen Hintergrundmessstationen, insbesondere für $PM_{10}$ (-5% in 2009, -9% in 2011 und -14% in 2013 an allen Hintergrundmessstationen, s. Abbildung 4). Dies würde auch erklären, warum die Wirksamkeit der Umweltzone in Augsburg an den Hintergrundmessstationen durch die Regressionsanalyse (wie in Kapitel 2 beschreiben) nicht nachzuweisen ist. In der Regression wurde die Messstation LfU als Referenzmessstation verwendet. Wenn die Minderung an dieser Messstation in der gleichen Größenordnung liegt wie an den anderen Hintergrundmessstationen in Augsburg, dann ist der Nachweis dieser Effekte in der Regression nicht möglich.

Um diesen Rückgang der Schadstoffkonzentrationen im Verhältnis zu der Gesamtbelastung darzustellen, muss eine Hintergrundbelastung in Augsburg angenommen werden. Hierbei sei zu beachten, dass die genannten Werte nicht zwingenderweise für alle Orte des Stadtgebietes in genau dieser zahlenmäßigen Größe gelten würden. Vielmehr seien regionale Schwankungen der Hintergrundbelastung um diese Werte anzunehmen.

Für die Wirkungsanalyse von Verkehrsbeschränkungen ist letztlich die absolute Höhe der Vorbelastung in erster Näherung nachrangig. Die Zielsetzung der Wirkungsanalyse besteht zu einem wesentlichen Teil darin, die erzielbaren Verbesserungen durch Verkehrsverbote bezogen auf einen definierten Bezugszustand zu errechnen. Dies gelingt daher auch unter Ansatz einer einheitlichen Hintergrundbelastung. Da die regionale Verteilung innerhalb der Umweltzone nicht bekannt ist, wird eine einheitliche Hintergrundbelastung von 15 µg/m³ für $PM_{10}$ und 20 µg/m³ für $NO_x$ zugrunde gelegt. Das würde bedeuten dass die erwarteten Änderungen der Gesamtbelastung durch die Einführung der Umweltzone in Augsburg bei etwa 1 bis 2% für $PM_{10}$ und unter 10% für $NO_x$ (für Hintergrundmessstationen) liegen.

In der Zeitperiode 2008 bis 2011 waren die Schwankungen der regionalen Belastung in der Größenordnung von 2,5 µg/m³ für $PM_{10}$ und 7,0 µg/m³ für $NO_x$ (Messstation Hohenpeißenberg, etwa 55 km südlich von Augsburg). Das bedeutet dass die erwarteten Effekte der Einführung der Umweltzone durch die Schwankungen des regionalen Beitrages (und der Meteorologie) überdeckt werden und nicht durch einen einfachen Vergleich der Jahresmittelwerte abgeschätzt werden können.

Abbildung 4: Berechnete Änderung der durch den lokalen Kfz-Verkehr verursachten Zusatzbelastung zu $NO_x$ und $PM_{10}$-Konzentrationen nach stufenweiser Einführung der Umweltzone in Augsburg (Meteorologie des Jahres 2008 wurde für alle nachfolgenden Jahre als konstant angenommen, es wurden nur die Beiträge des lokalen KfZ-Verkehrs verändert).




## 3.4 Vergleich zwischen IMMIS- und AirViro-Modell

Die Stadt Augsburg hat eine Wirkungsanalyse der Effektivität der Umweltzone durchführen lassen (Bayerisches Landesamt für Umwelt, 2010). Die Wirkungsanalyse wurde mit dem oben beschriebenen IMMIS$^{luft}$ Modell durchgeführt.

Die Durchführung der Berechnungen im IMMIS wurde anhand der konkreten Zulassungszahlen im Stadtgebiet Augsburg sowie in den umgebenden Landkreisen Augsburg sowie Aichach-Friedberg einerseits, und anhand der Verkehrszusammensetzung im Bundesdurchschnitt andererseits, vorgenommen. Die Untersuchung beschränkte sich hierbei auf die 17 am höchsten belasteten Straßenabschnitte in Augsburg. In unserer Analyse mit dem AirViro Modell haben wir im Gegenteil dazu nur Standorte analysiert, die im städtischen Hintergrund liegen und somit repräsentativer für die gesamte Stadtpopulation sind (90% der Einwohner wohnen im städtischen Hintergrund, 10% an stark belasteten Straßen und Kreuzungen).

In der vom Bayerischen Landesamt für Umwelt und Gesundheit in Auftrag gegebenen Studie wurde die Zusammensetzung der Fahrzeugflotte in Augsburg dargestellt. Der Vergleich der Zulassungszahlen der regionalen Fahrzeugflotte mit dem Bundesdurchschnitt zeigt, dass die lokale Fahrzeugflotte insgesamt deutlich jünger ist als der Bundesdurchschnitt. Bei den Pkws im Bundesdurchschnitt ist der Anteil an Fahrzeugen, welche keine Plakette erhalten würden, deutlich höher als in Augsburg und in der Region Augsburg. So hatten 2006 im Bundesdurchschnitt 26,2 % der Pkws keine Plakette erhalten, in 2010 waren noch 8,3 % der Pkws ohne Plakette. Im Vergleich hierzu lag der Prozentsatz der Pkws ohne Anspruch auf eine Plakette in Augsburg in 2010 bei 0,8 %. Dies ist vor allem auf die Einführung der Stufe 1 der Umweltzone in 2010 zurückzuführen. Der Anteil der Fahrzeuge ohne Plakette an der Flotte reduzierte sich in Augsburg von 2% auf 0,8 %. Insgesamt bedeutet das aber, dass der Effekt der Einführung der Umweltzone in Augsburg schon aus diesem Grund schwächer sein muss als zum Beispiel in Berlin, wo die Fahrzeugflotte deutlich älter ist.

Das errechnete Konzentrationsniveau der Summe aus Vor- und Zusatzbelastung bezüglich des Jahresmittelwerts für 2008 lag für $PM_{10}$ bei 26,3 µg/m³, wobei der Anteil der Kfz-abgasbedingten Zusatzbelastung durchschnittlich 2,3 µg/m³ betrug. In 2010 betrug das errechnete Konzentrationsniveau der Gesamtbelastung bezüglich des Jahresmittelwerts von $PM_{10}$ noch im Mittel 25,4 µg/m³, wobei der Anteil der Kfz-abgasbedingten Zusatzbelastung auf durchschnittlich 1,4 µg/m³ sank. Die Wirksamkeit der Maßnahme betrug somit etwa 0,9 µg/m³ im Durchschnitt bezogen auf 2008. Unter Ausblendung der allgemeinen Flottenverjüngung betrug die Wirksamkeit rund 0,3 µg/m³, was einer Reduktion der Kfz-abgasbedingten Zusatzbelastung um etwa 10% entspricht.

Diese Zahlen sind gut mit dem Ergebnissen des AirViro-Modells vergleichbar (Reduktion der Kfz-abgasbedingten Zusatzbelastung um 0,1 µg/m³ bei der Einführung der ersten Stufe und bis 0.3 µg/m³ bei der Einführung der dritten Stufe der Umweltzone), wenn man berücksichtigt, dass bei der Analyse mit dem IMMIS-Modell hochbelastete Straßenstandorte untersucht wurden, und in der Analyse mit dem AirViro-Modell Hintergrundstandorte untersucht wurden.

# 4 Abschätzung der personenspezifischen Exposition

In epidemiologischen Kurzzeitstudien werden überwiegend die Außenluftkonzentrationen des Feinstaubs in der Regel an fest eingerichteten Messstellen ermittelt und den Bewohnern des jeweiligen (oft mehrere Quadratkilometer großen) Einzugsgebiets zugeordnet. Einige Studien haben gezeigt, dass die Korrelationen zwischen personenspezifischen Luftschadstoffkonzentrationen und Konzentrationen die an einer zentralen Messstation gemessen wurden (Außenluftkonzentrationen), relativ schwach sind (Oglesby et al., 2000, Meng et al., 2005). Das bedeutet, dass die Außenluftkonzentrationen nur bedingt die personenspezifische Exposition abbilden können. Die so entstehenden, nicht systematischen Unschärfen der Expositionsbestimmung (sog. „nondifferential misclassification“) wirken sich häufig so aus, dass statistische Zusammenhänge zwischen Exposition und gesundheitlichen Effekten abgeschwächt werden. Dies ist umso stärker ausgeprägt, je inhomogener ein Schadstoff über ein Studiengebiet verteilt ist, also insbesondere für ultrafeinen Partikel (UFP) oder Dieselruß. Mit Expositionsmodellen, die sich auf Messungen in unterschiedlichen Mikrokompartimenten und die Aufenthaltszeiten in diesen Lebenswelten stützen, wurde versucht, die „Gesamt“-Exposition und den Beitrag der einzelnen Kompartimente besser abzuschätzen. Ein solches Modell sollte auch im Rahmen des ACCEPTED-Projekts entwickelt werden.

Die Universität Augsburg hat sich verpflichtet, bereits vorhandene Daten zur personenspezifischen Exposition gegenüber UFP, kohlenstoffhaltigen Partikeln (Black Carbon - BC) sowie $PM_{2,5}$ auszuwerten und dem Projektpartner AIRPARIF (Surveillance de la qualité de l'air en Île-de-France (air quality monitoring network), Paris) zur Validierung der Modelle zur Verfügung zu stellen. Die Daten wurden im Rahmen einer durch die Deutsche Forschungsgemeinschaft finanzierten Studie „Combining individual and central site measurements of ultrafine particles: Complex statistical analyses of source-dependent health effects“ durch die Wissenschaftler der Universität Augsburg erhoben (Deffner und Kollegen, 2015). Die ultrafeinen Partikel wurden in dieser Studie durch Erfassung der Partikelanzahlkonzentration (PNC = particle number concentration) bestimmt.

Die Schadstoffe wurden in unterschiedlichen Szenarien gemessen: entlang der Hauptstraße, in einem Wohn- und einem Industriegebiet sowie in Bussen und Straßenbahnen. Die Messungen wurden in drei unterschiedlichen Jahreszeiten (Winter, Frühling und Sommer) jeweils in einer Zeitperiode von drei Wochen durchgeführt. Die personenspezifischen Messungen von PNC wurden mit einem Kondensations-Partikel Zähler (CPC = Condensation Particle Counter, TSI Inc. USA, Model 3007) durchgeführt. Für die Messungen von BC als Marker für Dieselruß wurde ein Äthalometer (Magee Scientific Inc., microAeth® Model AE51) und für Messungen von $PM_{2,5}$ ein personengetragener Aerosolmonitor (Thermo Inc. USA, Model DataRAM pDR-1500) verwendet. Des Weiteren wurden PNC, BC und $PM_{2,5}$ Daten an einer zentralen Aerosol-Messstation in Augsburg gesammelt. Diese Messstation befindet sich auf dem Gelände der Hochschule Augsburg, ca. 1 km südöstlich von der Stadtmitte entfernt und wird als eine städtische Hintergrundstation charakterisiert (Cyrys et al., 2008). Vor jeder Messrunde wurden die personengetragenen Messgeräte zur Erfassung von PNC, BC und $PM_{2,5}$ an der Messstation mit den stationären Geräten verglichen.

Der Untersucher ging jeden Tag von Montag bis Freitag entlang der gleichen Route: entlang einer Hauptverkehrsstraße, in einem Wohngebiet, in einem Industriegebiet, in einem Wohngebiet mit erhöhtem Anteil an Holzverbrennung und in Bussen und Straßenbahnen zwischen den unterschiedlichen Gebieten (Abbildung 5). Die Messung begann normalerweise morgens ungefähr um 8:00 Uhr und dauerte durchschnittlich ca. 5,5 Stunden an. Die Route wurde täglich durch GPS Datenlogger festgehalten. Tabelle 8 zeigt die durchschnittliche Aufenthaltsdauer in den einzelnen Mikrokompartimenten.

Abbildung 5: Karte von Augsburg mit eingezeichnetem Verlauf der Untersuchungsstrecke für die Bestimmung von personenspezifischen Konzentrationen von UFP, BC und $PM_{2,5}$.

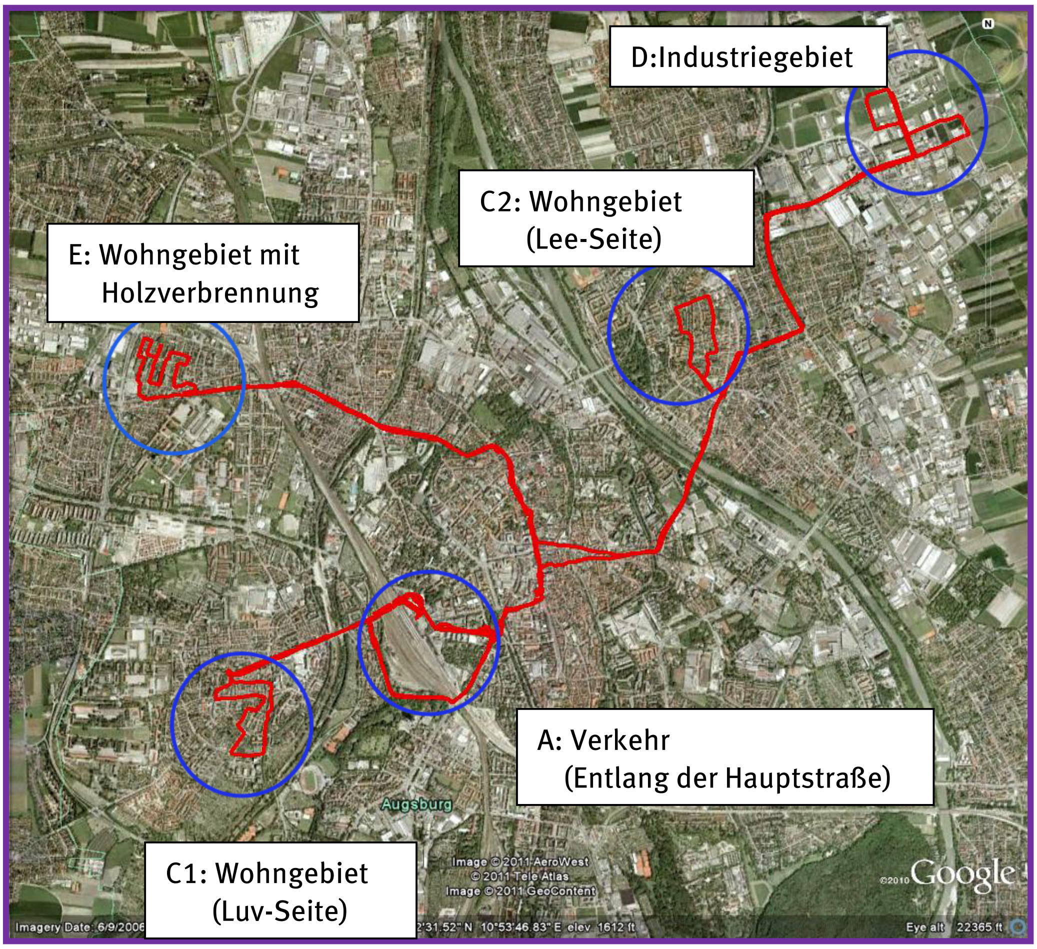



Tabelle 8: Durchschnittliche Aufenthaltsdauer in den einzelnen Mikrokompartimenten.

| Bezeichnung | Mikrokompartiment | Dauer | Anfangszeit |
|---|---|---|---|
| A1 | Entlang der Hauptstraße vormittags | ~30 min | 8:00 |
| B1 | In der Straßenbahn | ~10 min | 8:40 |
| C1 | Wohngebiet (Luv-Seite) | ~30 min | 8:55 |
| B1 | In der Straßenbahn | ~21 min | 9:25 |
| C2 | Wohngebiet (Lee-Seite) | ~30 min | 9:55 |
| B2 | Im Bus | ~10 min | 10:25 |
| D | Industriegebiet | ~30 min | 10:40 |
| B1/B2 | Im Bus und in der Straßenbahn | ~39 min | 11:25 |
| E | Wohngebiet mit Holzverbrennung | ~30 min | 12:00 |
| B | In der Straßenbahn | ~30 min | 12:30 |
| A2 | Entlang der Hauptstraße mittags | ~30 min | 13:00 |

Die personenspezifischen Luftschadstoffkonzentrationen im Frühjahr und im Sommer waren sehr ähnlich. Für die nachfolgende Auswertung wurden sie deshalb zusammengefasst und sind in den Abbildung 6 und Abbildung 7 als Kategorie „warm“ zu sehen. Die Ergebnisse der Messungen im Winter sind als Kategorie „kalt“ dargestellt. In Abbildung 6 sind die personenspezifischen $PM_{2,5}$, BC und PNC Konzentrationen für die Kategorie „kalt“ (Winter) und „warm“ (Frühjahr und Sommer zusammen) dargestellt.

Generell sind die Konzentrationen in der kalten Jahreszeit höher als in der warmen Jahreszeit. Auffallend ist die besonders hohe $PM_{2,5}$ Konzentration im Winter in Wohngebieten (C1, C2, D) verglichen mit den Konzentrationen, die in der warmen Jahreszeit gemessen wurden. Dies könnte durch die Verbrennung von fossilen Brennstoffen für Hausheizung, unter anderem auch Holzverbrennung, verursacht sein. Die höchste personenspezifische $PM_{2,5}$ Konzentration wurde im Bus gemessen (B2), während die Exposition in der Straßenbahn niedriger war (B1). Die Konzentration im Bus war sogar höher als die während des Aufenthalts an einer verkehrsreichen Straße (A1 bzw. A2).

An der Hauptstraße wurden hohe personenspezifischen Expositionen gegenüber ultrafeinen Partikeln sowohl in der kalten Jahreszeit (63 302 $cm^{-3}$) als auch in der warmen Jahreszeit (37 361 $cm^{-3}$) beobachtet. Im Winter wurden ebenfalls sehr hohe BC Konzentrationen von 15 $\mu g/m^3$ an der Hauptstraße morgens (A1) und im Bus (B2) beobachtet.

Das Verhältnis der personenspezifischen Luftschadstoffkonzentrationen zu der Konzentration in der Außenluft, die an einer Referenzmessstation im städtischen Hintergrund gemessen wurde (p/A Verhältnis) gibt Hinweise auf den Beitrag der unterschiedlichen Außenluft-Quellen zur personenspezifischen Exposition. In der Abbildung 7 sind die p/A Verhältnisse dargestellt. Für die Wohngebiete war die Erhöhung der personenspezifischen Exposition für alle Schadstoffe generell weniger ausgeprägt als an den anderen Aufenthaltsorten und überstieg nicht den Faktor 2. Das bedeutet, dass die Außenluftmessungen im städtischen Hintergrund repräsentativ für Menschen sind, die sich überwiegend im städtischen Hintergrund aufhalten. In Verkehrssituationen (Aufenthalt an der Verkehrsstraße, Aufenthalt im Bus oder in der Straßenbahn) wurden deutlich erhöhte personenspezifischen Konzentrationen an Luftschadstoffen gemessen. Die Erhöhung war besonders stark für BC, gefolgt von PNC. So war die personenspezifische BC Konzentration, die im Bus (B2) gemessen wurde, bis sechsmal höher als die Konzentrationen in der Außenluft. Deutlich erhöht war auch die personenspezifische BC Konzentration für den Aufenthalt an der verkehrsreichen Straße, im Sommer sogar mehr als im Winter. Die Anreicherung von $PM_{2,5}$ war weniger stark ausgeprägt.

Abbildung 6: Mittlere personenspezifische Konzentrationen von $PM_{2,5}$, BC, und PNC in unterschiedlichen Aufenthaltsorten.

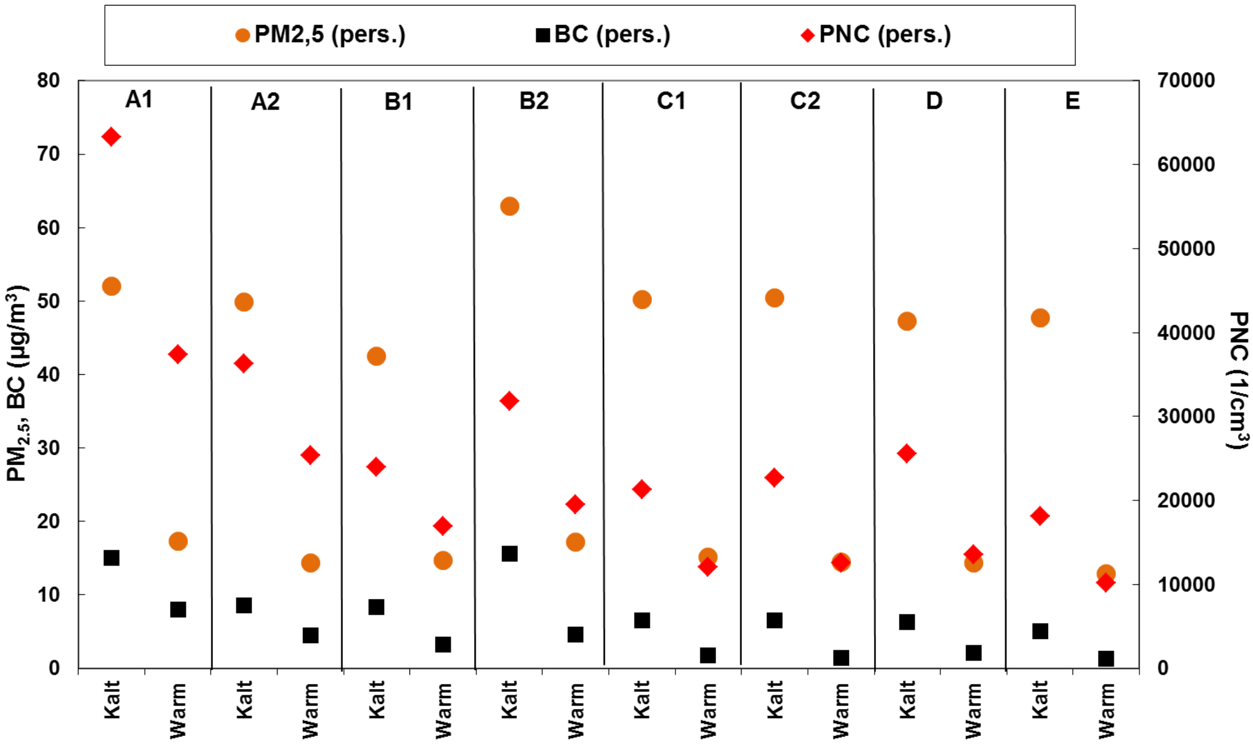
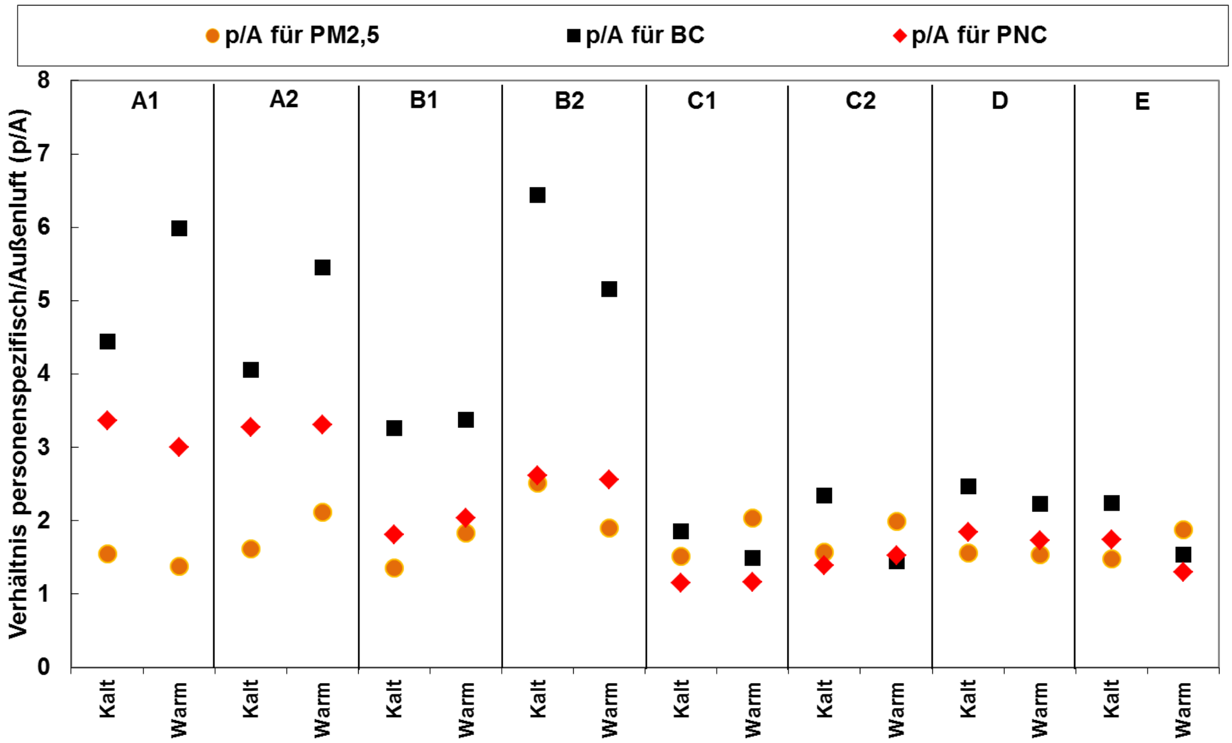



Abbildung 7: Mittlere Verhältnisse zwischen den personenspezifischen Konzentrationen von $PM_{2,5}$, BC, und PNC und den Außenluftkonzentrationen (Referenzmessstation).



Die Spearman Korrelationskoeffizienten zwischen den individuellen Konzentrationen und den Außenluftkonzentrationen für die zeitlich hoch aufgelösten Messungen in minütlicher Auflösung betragen 0,87 für $PM_{2,5}$, 0,48 für BC und 0,39 für PNC. Die gleichen Korrelationskoeffizienten für die Tagesmittelwerte sind deutlich höher (r=0,94 für $PM_{2,5}$ und BC und 0,81 für PNC). Die Streudiagramme (Abbildung 8 und 9) machen die Unterschiede zwischen der Korrelation zwischen den zeitlich hoch aufgelösten Daten (hier in stündlicher Auflösung) und der Korrelation zwischen den Tagesmittelwerten deutlich.

Abbildung 8: Streudiagramm zum Verhältnis von personenspezifischer Exposition gegenüber $PM_{2,5}$ und der $PM_{2,5}$ Konzentration an der Referenzmessstation (grün=Frühjahr, rot=Sommer, blau=Winter).
Linke Abbildung: Stundenmittelwerte; rechte Abbildung: Tagesmittelwerte.

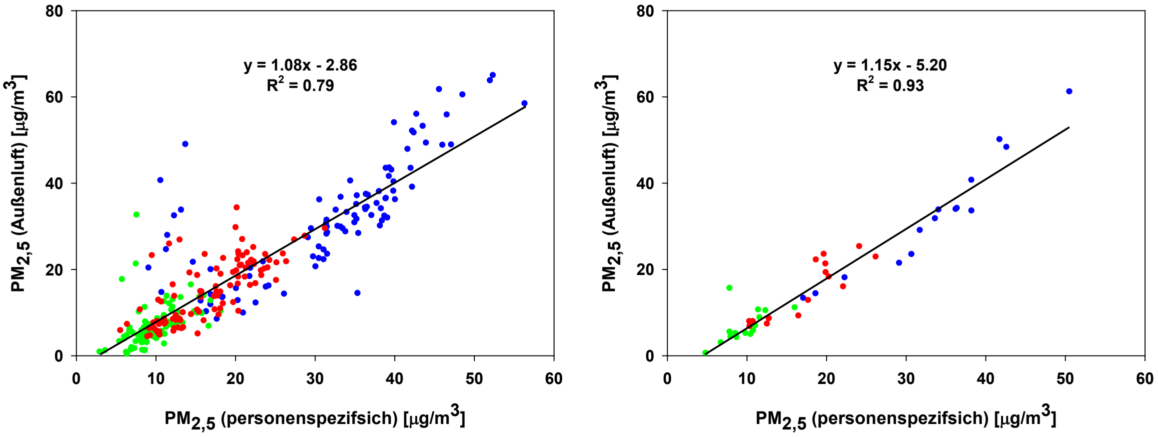
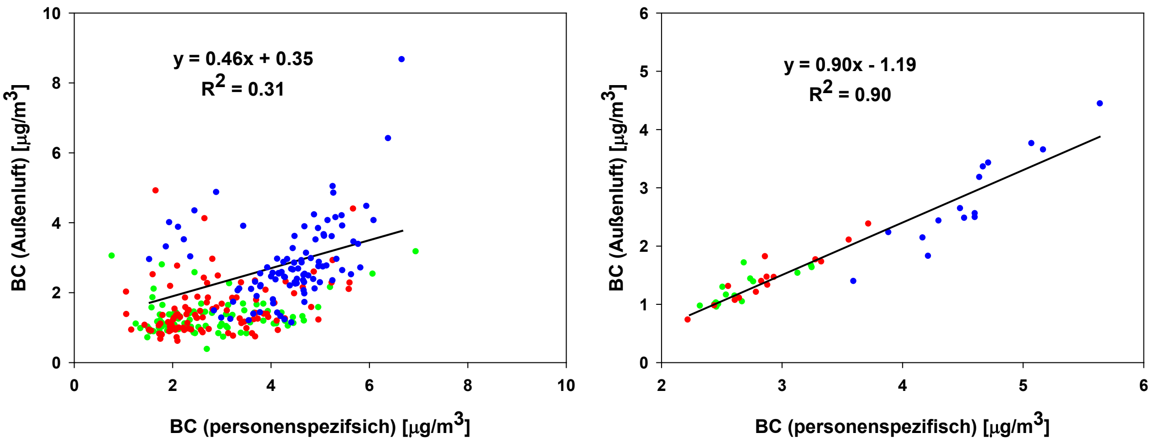



Abbildung 9: Streudiagramm zum Verhältnis von personenspezifischer Exposition gegenüber BC und der BC Konzentration an der Referenzmessstation (grün=Frühjahr, rot=Sommer, blau=Winter).
Linke Abbildung: Stundenmittelwerte; rechte Abbildung: Tagesmittelwerte.




Die Ergebnisse sind relevant für die Interpretation epidemiologischer Kurzzeitstudien. Im Allgemeinen findet man relativ niedrige Korrelationen zwischen Außenluftkonzentrationen und gemessenen Konzentrationen im Innenraum auf der einen Seite und den Messergebnissen des personenspezifischen Monitorings auf der anderen Seite. Die von uns beobachteten hohen Korrelationen zwischen Außenluftkonzentrationen und personenspezifischer Exposition bei Personen ohne Innenraumquellen deuten darauf hin, dass die in anderen Studien nachgewiesenen deutlich schwächeren Zusammenhänge offenbar an den großen individuellen Unterschieden des Beitrags von nicht-Außenluftquellen abhängen. So kann man hiernach die zunächst paradox erscheinende Tatsache verstehen, dass statistisch signifikante Zusammenhänge zwischen Gesundheitseffekten und Außenluftkonzentrationen gefunden werden, obwohl die Menschen sich überwiegend in Innenräumen aufhalten und die Korrelationen zwischen Außenluft- und Innenraumkonzentrationen sehr niedrig sind: Außenluft- und Innenraumquellen sind praktisch nicht korreliert, und daher sind Partikel aus der Außenluft und im Innenraum als unabhängige Schadstoffe anzusehen, deren Auswirkungen man getrennt schätzen kann.

# 5 Veröffentlichung der Ergebnisse, Teilnahme an Konferenzen

In diesem Bericht wurden nur die Arbeiten der Universität Augsburg beschrieben, die im Rahmen des Projekts ACCEPTED (Assessment of changing conditions, environmental policies, time-activities, exposure and disease) durchgeführt worden sind. Eine Darstellung der Ergebnisse des gesamten Projekts ist in dem englischsprachigen Abschlussbericht zu finden. In Anlage IA ist die deutsche Zusammenfassung und in IB die englische Zusammenfassung des Berichts zu finden. Der gesamte Endbericht ist in Anlage II sowie auf der Internetseite des ACCEPTED Projekts zu finden (http://www.acceptedera.eu) und wurde zusammen mit diesem Bericht in elektronischer Form eingereicht (Dateiname: ACCEPTED Final Report.pdf).

Die Ergebnisse des Projekts wurden auch in Teilberichten publiziert. Bis jetzt wurden zwei Teilberichte erstellt, die ebenfalls auf der Internetseite des Projekts zu finden sind (http://www.acceptedera.eu/members/important-documents/). Es handelt sich um folgende Teilberichte:

Johansson C., Andersson C., Bennet C., Gidhagen L., Cyrys, J., Soentgen J., Gu J.: “Report on Low Emission and Congestion Charge Zones in Europe with impact assessment studies for Augsburg, Munich, Berlin and Stockholm”. ACCEPTED report, June 2014.

Andersson C., Johansson C., Markakis K., Valari M., Delcloo A., Hamdi R.: “Report on the effect of future climate and control policies on air quality in Stockholm, Paris and Brussels”. ACCEPTED report, May 2015.

Die Universität Augsburg war maßgeblich an der Erstellung des ACCEPTED Teilberichts über die Analyse der Effekte von Umweltzonen beteiligt (Johannsson et al., 2014). Dieser Teilbericht ist in Anlage III und auf der Internetseite des ACCEPTED Projekts zu finden (http://www.acceptedera.eu). Er wurde auch zusammen mit diesem Bericht in elektronischer Form eingereicht (Dateiname: LEZ_report_2014_June19.pdf).

Darüber hinaus wurden die Ergebnisse des Projekts auf Konferenzen und Tagungen sowie durch Veröffentlichungen vorgestellt und publiziert.

## 5.1 Weitere Veröffentlichungen

Cyrys, J., Peters, A., Soentgen, J., Wichmann, H.-Erich: Low Emission Zones Reduce $PM_{10}$ Mass Concentrations and Diesel Soot in German Cities. Journal of the Air & Waste Management Association 2014, 64 (4): 481 – 487.

Fensterer, V., Küchenhoff, H., Maier, V., Wichmann, H.E., Breitner, S., Peters, A., Gu, J., Cyrys, J.: Evaluation of the Impact of Low Emission Zone and Heavy Traffic Ban in Munich (Germany) on the Reduction of $PM_{10}$ in Ambient Air. International Journal of Environmental Research and Public Health 2014, 11, 5094 – 5112.

Morelli, X.C., Rieux, C., Cyrys, J., Forsberg, B., Slama, R.: Air pollution, health and social deprivation: a fine-scale risk assessment. Environmental Research 2016, 147, 59 – 70.

## 5.2 Buchkapitel

Cyrys, J., Peters, A., Soentgen, J., Gu, J., Wichmann, H.-E.: Umweltzonen. In Handbuch der Umweltmedizin – 53. Erg. Lfg. 12/14, 2014.

## 5.3 Veröffentlichungen in Tagungsbändern

Cyrys, J., Fensterer, V., Küchenhoff, H., Bauer, B., Wichmann, H.-E., Breitner, S., Schneider, A., Peters, A.: The Impact of Low Emission Zone and Heavy Traffic Ban in Munich on the Reduction of $PM_{10}$ in Ambient Air. Proceedings of Abstracts 9th International Conference on Air Quality, Science and Application, Garmisch-Partenkirchen, 24-28 March 2014

Cyrys, J., Gu, J., Deffner, V., Küchenhoff, H., Soentgen, J., Peters, A.: Analyse der Wirksamkeit von Umweltzonen in drei deutschen Städten: Berlin, München und Augsburg. In VDI Berichte 2250 "Neue Entwicklungen bei der Messung und Beurteilung der Luftqualität". Nürnberg, 20.-21. Oktober 2015.

## 5.4 Vorträge auf Konferenzen und Tagungen

Cyrys, J., Gu, J., Dörsch, M., Bauer, B., Küchenhoff, H., Fensterer, F.: "LEZ in München & Augsburg - Analyses of impacts based on measurements of $PM_{10}$". ACCEPTED Workshop "Low Emission Zones – how to estimate the air quality benefits". Stockholm, 17-18 Juni, 2014.

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Cyrys, J., Fensterer, V., Küchenhoff, H., Bauer, B., Wichmann, H.-E., Breitner, S., Schneider, A., Peters, A.: The Impact of Low Emission Zone and Heavy Traffic Ban in Munich on the Reduction of $PM_{10}$ in Ambient Air. 9th International Conference on Air Quality, Science and Application, Garmisch-Partenkirchen, 24-28 March 2014

Cyrys, J., Gu, J., Deffner, V., Küchenhoff, H., Soentgen, J., Peters, A.: Effects of Low Emission Zones (LEZ) on air quality in Germany. European Aerosol Conference 2015, Milan, September 6 -11, 2015.

Cyrys, J., Gu, J., Deffner, V., Küchenhoff, H., Soentgen, J., Peters, A.: Analyse der Wirksamkeit von Umweltzonen in drei deutschen Städten: Berlin, München und Augsburg. VDI seminar "Neue Entwicklungen bei der Messung und Beurteilung der Luftqualität". Nürnberg, 20.-21. Oktober 2015.

Johansson, C., Cyrys, J., Soentgen, J., Gu, J., Andersson, C., Bennet, C., Gidhagen, L.: Air pollution exposure in European cities: Effects of traffic policy. ACCEPTED final meeting. 23 October 2015, Brussels.

Morelli, X.C., Rieux, C., Cyrys, J., Forsberg, B., Slama, R.: Air pollution, health and social deprivation in an urban setting: a fine-scale risk assessment. ACCEPTED final meeting. 23 October 2015, Brussels.

## 5.5 Posterpräsentationen auf Konferenzen und Tagungen

Cyrys, J., Gu, J., Deffner, V., Küchenhoff, H., Maier, V., Soentgen, J., Peters, A.: Effects of Low Emission Zone in three German cities. ACCEPTED final meeting. 23 October 2015, Brussels.

Gu, J., Andersson, C., Bennet, C., Gidhagen, L., Wolf, K., Schneider, A., Soentgen, J., Peters, A., Cyrys, J.: Modelling of Exposure to Ambient Particulate Matter and $NO_x$ in Augsburg, Germany. European Aerosol Conference 2015, Milan, September 6 -11, 2015.

## 5.6 Eingereichte Veröffentlichungen oder in Vorbereitung

Gu, J., Deffner, V., Küchenhoff, H., Slama, R., Morelli, X., Breitner, S., Schneider, A., Wichmann, HE., Peters, A., Cyrys, J.: The effects of Low Emission Zone in reducing particulate matter pollution in three German cities. (In Vorbereitung)

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# 7 Anlagen

**Anlage IA: Zusammenfassung des ACCEPTED Endberichts**

Änderungen in Stadtplanung und Verkehrspolitik, demographischer Wandel, Klimawandel und Umweltpolitik werden wahrscheinlich die Qualität der Innenraum- und Außenluft und damit auch die öffentliche Gesundheit verändern. ACCEPTED ist ein interdisziplinär angelegtes Forschungsprojekt mit dem Ziel, zukünftige Expositionsszenarien in Städten und ihren Einfluss auf die menschliche Gesundheit besser zu verstehen. Hierfür wurden modernste Klimamodelle sowie Messungen verwendet, die Trends und mögliche Änderungen der Exposition beschreiben, und auf epidemiologische Expositions-Wirkungsfunktionen anwenden.

Der Projektname "ACCEPTED" ist ein Akronym für "Assessment of Changing Conditions, Environmental Policies, Time-activities, Exposure and Disease". Das Projekt begann im Dezember 2012 und endete im Dezember 2015. Die Internetseite des Projekts (http://www.acceptedera.eu) bleibt jedoch auch nach Beendigung des Projekts aktiv um Publikationen hervorzuheben, die zum Zeitpunkt des Projektendes noch nicht publiziert waren (siehe auch die Liste der eingereichten Manuskripte im Literaturverzeichnis dieses Abschlussberichts).

Zweimal jährlich (2013 bis 2015) wurden Neuigkeiten von ACCEPTED in einem Projekt-Newsletter vorgestellt. Alle Ausgaben der Newsletter können auf der Projekt-Homepage eingesehen werden.

ACCEPTED umfasst 11 unterschiedliche Partner und wurde von fünf nationalen Organisationen (ADEME und ANSES in Frankreich, BelSPO in Belgien, der schwedischen Umweltbehörde EPA in Schweden und UBA in Deutschland) im Rahmen des ERA-ENVHEALTH Netzwerks finanziert.

Projektpartner und Kontaktpersonen sind:

1. Umeå University (Projektkoordinator), Bertil Forsberg
1. Swedish Meteorological and Hydrological Institute (SMHI), Camilla Andersson
2. City of Stockholm, Christer Johansson
3. Hasselt University, Tim Nawrot
4. Royal Meteorological Institute of Belgium (RMI), Rafiq Hamdy
5. Universität Augsburg, Josef Cyrys
6. Centre National de la Recherche Scientifique (CNRS), Myrto Valari
7. Institut national de la santé et de la recherche médicale (INSERM), Rémy Slama
8. Centre Scientifique et Technique du Bâtiment (CSTB), Bernard Collignan, Emilie Powaga
9. AIRPARIF, Cecile Honoré
10. Institut scientifique de service public (ISSeP), Suzanne Remy

**Wirkung verminderter Emissionen und Klimaänderungen**

Wir erwarten, ausgehend von einem Ensemble aus 4 Modellen, dass eine Reduktion der europäischen anthropogenen Emissionen bis 2050 zu einer deutlichen Verbesserung der Luftqualität führen wird und dabei generell einen stärkeren Einfluss auf die Änderung der Luftqualität in Europa haben wird, als der Klimawandel. Diese Schlussfolgerung ist jedoch abhängig von der räumlichen Auflösung des Emissionskatasters und der angewendeten Modelle. Wenn höher aufgelöste Modelle für Paris angewendet werden, erweist sich der Klimawandel als stärkster Faktor für Änderungen der maximalen Ozonkonzentration, während die Partikel-Konzentrationen und die mittlere Ozonkonzentration vor allem durch lokale Emissionen beeinflusst werden. Das deutet darauf hin, dass eine grobe räumliche Auflösung der Emissionsdaten den Effekt der zukünftigen Maßnahmen zur Emissionsreduktion auf die Senkung der Ozon-Gesamtkonzentration überschätzen kann. Folglich kann die Anwendung von grob aufgelösten Emissionskatastern für die Modellierung zu signifikanten Fehlern bei der Beurteilung von Maßnahmen führen, selbst in Regionen mit starkem regionalem Anteil der Luftschadstoffe wie in Stockholm. Wir erwarten, dass aufgrund der Klimaänderung bis Mitte des 21. Jahrhunderts eine Abnahme der Ozon-Konzentration in Stockholm, eine Abnahme der Partikel- und Ozon-Konzentrationen in Paris, sowie eine Zunahme der Partikel-Konzentration in Stockholm und Brüssel erfolgen wird.

Die Modellierung der Klimaänderungen im städtischen Maßstab bis zum Jahr 2050 zeigt eine beträchtliche Erwärmung um bis zu 1,6°C für Brüssel und bis zu 1,8°C für Paris. Die Ergebnisse lassen eine Verringerung der Intensität von städtischen Hitzeinseln (UHI, urban heat islands) während des Tages im Sommer erkennen, die durch Austrocknung von Böden in ländlichen Gebieten bedingt ist, sowie eine Zunahme der nächtlichen UHI-Intensität im Winter, die durch abnehmende Windgeschwindigkeit verursacht wird. Die erwartete Zunahme der nächtlichen UHI-Intensität im Winter wird eine Zunahme der Partikel-Konzentrationen in den untersuchten Studiengebieten verursachen. Da die Zunahme der nächtlichen UHI-Intensität insbesondere Gebiete betreffen werden, die sich durch eine hohe Populationsdichte und hohe anthropogene Luftschadstoffemissionen auszeichnen, ist es für politische Entscheidungsträger besonders wichtig sicherzustellen, dass die anthropogenen Emissionen weiterhin reduziert werden, um dem negativen Einfluss zukünftiger Klimaänderungen entgegenzuwirken. Die Ergebnisse wurden unter Anwendung neuer Downscaling-Modellierungstechniken für Stadtklima erzielt, die im Rahmen dieses Projekts entwickelt wurden. Es wird erwartet, dass die Entwicklung der Städte und die in Zukunft häufigeren extremen Klimaereignisse (wie zum Beispiel Hitzewellen) einen starken Einfluss auf städtische Hitzeinseln haben werden.

**Nutzen lokaler Minderungsmaßnahmen**

Für Stockholm quantifizierten wir mit Hilfe von Szenarien den Einfluss verschiedener Minderungsmaßnahmen auf die zukünftige Luftqualität. Eine Verlagerung des Autoverkehrs von einer Hauptstraße in einen Tunnel in Stockholm würde die negativen gesundheitlichen Effekte für die Anwohner aufgrund der Expositions-Reduktion gegenüber verkehrsabhängigen Luftschadstoffen mindern. Dieser positive Effekt würde jedoch durch eine höhere Exposition der Autofahrer im Tunnel aufgehoben. Ferner zeigten wir, dass eine Erhöhung der City-Maut nur einen geringen Effekt auf die Exposition haben würde. Der Austausch mit fossilen Brennstoffen betriebener Fahrzeuge durch Elektroautos würde zwar die Partikel-Exposition reduzieren, die Ozon-Exposition aber gleichzeitig erhöhen. Allerdings erwarten wir, dass die positive Wirkung der Klimaänderung auf die bodennahen Ozon-Konzentrationen in Stockholm stärker wird. Die europaweite Verringerung der Emissionen bis 2050 wird eine starke Reduktion sowohl der Partikelmasse als auch Ozon-Konzentrationen verursachen, die den Einfluss der Klimaänderung bzw. lokaler Maßnahmen bei weitem übersteigen wird. Allerdings würde ein Verbot von Spikereifen zu einer signifikanten Reduktion der $PM_{10}$-Konzentrationen führen, die etwa 50% der Minderung ausmacht, die aufgrund der europaweiten Emissionsänderungen bis 2050 zu erwarten ist.

Für eine Bewertung von bereits eingeführten Minderungsmaßnahmen haben wir den Schwerpunkt insbesondere auf den Einfluss der Einführung von Umweltzonen in München, Berlin und Augsburg sowie einer City-Maut Zone in Stockholm gelegt. Wir zeigten, dass die Effektivität der Umweltzone von der Schärfe der Regelungen sowie ihrer Größe abhängig ist und dass sie sich besonders effektiv hinsichtlich der Reduktion der gesundheitsrelevanten Bestandteile (wie zum Beispiel Dieselruß) in $PM_{10}$ erweisen. Das bedeutet, dass auch wenn der Einfluss der Umweltzonen auf $PM_{10}$ gering sein mag, der Einfluss auf die Gesundheit stärker ist, da die Umweltzonen insbesondere die Rußpartikel reduzieren, die in Verdacht stehen, die stärksten negativen Gesundheitseffekte zu haben. Um den Einfluss dieser Maßnahmen angemessen beurteilen zu können, ist es wichtig, die Verkehrsdichte und die Zusammensetzung der Kfz-Flotte innerhalb und außerhalb der Umweltzone zu erheben und zu bewerten. Darüber hinaus wäre die Messung von elementarem Kohlenstoff (EC, elemental carbon) und/oder organischem Kohlenstoff (OC, organic carbon) in $PM_{10}$ besser geeignet, um die Wirksamkeit von Umweltzonen aufzuzeigen.

**Exposition in der Außenluft und in Innenräumen**

Gemäß einem Modell für den Großraum Paris trägt die Exposition in Innenräumen zu 90% der gesamten Populationsexposition gegenüber bodennahem Ozon bei und zu 87% gegenüber $PM_{2,5}$. Die Exposition gegenüber Ozon ist generell höher für Anwohner in ländlichem Gebiet. Weil die Häuser in der Innenstadt von Paris älter sind als die im ländlichen Raum, was zu einer höheren Infiltrationsrate von Ozon führt, hat die Erhöhung der städtischen Außenluftkonzentrationen einen relativ geringen Effekt auf die personenspezifische Exposition. Menschen im Ruhestand, eine der vulnerablen Gruppen im ländlichen Raum, die in alten Häusern wohnen, sind 40% höheren Konzentrationen ausgesetzt als die durchschnittliche Exposition der Bevölkerung.

Eine bessere Wärmedämmung zur Energieersparnis wird zu einer geringeren Exposition gegenüber bodennahem Ozon führen, während dieser Effekt auf $PM_{2,5}$ schwach ist.

**Gesundheitseffekte bei vulnerablen Gruppen**

Im Großraum Stockholm wurde eine große Studie durchgeführt, um den Zusammenhang zwischen verkehrsabhängigen Luftschadstoffen an der Wohnungsadresse und Inzidenz von Frühgeburten sowie schwangerschaftsbedingtem Bluthochdruck zu untersuchen. Es wurde ein Zusammenhang zwischen erhöhter Exposition gegenüber verkehrsabhängigen Luftschadstoffen und schwangerschaftsbedingtem Bluthochdruck festgestellt. Eine Erhöhung der NOx-Konzentration in der Außenluft an der Wohnungsadresse während der Schwangerschaft um 10 µg/m3 erhöhte das relative Risiko um 17,0% (95% Konfidenz-intervall: 10,0-26,0). Auch das relative Risiko für Frühgeburten war bei erhöhten NOx-Konzentrationen tendenziell erhöht. Im zweiten Teil der Studie wurden Assoziationen zwischen verkehrsbedingten Partikeln und Gesundheit untersucht. Es zeigten sich ähnliche Zusammenhänge: erhöhte Konzentrationen der verkehrsbedingten Partikel gingen mit Schwangerschaftstoxikose, erhöhtem Blutdruck während der Schwangerschaft sowie der Inzidenz von Frühgeburten einher.

Ferner wurden in einer anderen Studie in Belgien Zusammenhänge zwischen Feinstaub-Exposition während der Schwangerschaft und Änderungen der Konzentrationen von Metaboliten in Nabelschnurblut festgestellt. Dies deutet auf eine Änderung des Entzündungsstatus von Neugeborenen hin, die durch Luftschadstoffe während des intrauterinen Wachstums verursacht werden.

Die Rolle von extremen Temperaturen als Auslöser von Frühgeburten in der warmen und kalten Jahreszeit wurde in Flandern (Belgien) und in Stockholm (Schweden) untersucht. Bezogen auf den Medianwert von 12,8 °C, war in Flandern das Risiko einer Frühgeburt in der warmen Jahreszeit um 14,1 % (95% Konfidenzintervall: 4,7-24,2) erhöht, wenn die minimale Temperatur an bis zu drei Tagen vor der Geburt über dem 95-ten Perzentil lag. Dieser Zusammenhang war weniger deutlich in Stockholm. Dort wurde ein Anstieg des relativen Risikos für eine Frühgeburt um 4-5 % festgestellt, wenn die mittlere Temperatur vier Wochen vor der Geburt über dem 75-ten Perzentil lag. Die Zusammenhänge mit extremeren Temperaturen waren nicht konsistent.

### Abschätzung der lokalen und europaweiten Gesundheitsfolgen

Studien zur Risikobewertung berücksichtigen oft nicht die räumliche Variabilität der Luftschadstoffe innerhalb einer Stadt. Unser Ziel war es, das attributable Risiko von Feinstaub ($PM_{2,5}$) in zwei Ballungsgebieten mit Hilfe von Dispersionsmodellierung zu quantifizieren und zu beschreiben wie dieser Effekt in Abhängigkeit von sozialer Benachteiligung variiert. Für den Großraum Grenoble und Lyon (0,4 und 1,2 Millionen Einwohner) wurde für 2012 die Exposition gegenüber $PM_{2,5}$ in einem 10 m x 10 m Raster durch Verknüpfung eines Dispersionsmodells mit der Populationsdichte abgeschätzt. Gesundheitsendpunkte waren die Inzidenzraten von Mortalität, Lungenkrebs und niedrigem Geburtsgewicht bei termingerechter Entbindung. Die Anzahl der Krankheitsfälle, die den Luftschadstoffen zugeschrieben werden können, wurde für die Gesamtfläche sowie stratifiziert nach Gebieten gemäß dem Europäischen Benachteiligungsindex (EDI: European Deprivation Index) geschätzt, indem die tatsächliche Situation mit dem von der Weltgesundheitsorganisation vorgeschriebenen Richtwert (10 µg/m$^3$ für den $PM_{2,5}$ Jahresmittelwert) als (alternative) Referenz verglichen wurde. Die Schätzungen wurden mit der Annahme einer homogenen Verteilung der Luftschadstoffe über das Stadtgebiet wiederholt. Die Medianwerte der $PM_{2,5}$ Konzentrationen lagen bei 18,1 in Grenoble und 19,6 µg/m$^3$ in Lyon, und gehen einher mit 114 (5,1% der Gesamtmortalität, 95% Konfidenzintervall: 3,2–7,0%) und 491 (6,0% der Gesamtmortalität, 95% Konfidenzintervall: 3,7–8,3%) vorzeitigen Todesfällen, die auf eine Langzeitexposition gegenüber $PM_{2,5}$ zurück geführt werden können.

Die attributablen Fälle von niedrigem Geburtsgewicht stellen 23,6% aller Fälle (9,0–37,1%) in Grenoble und 27,6% aller Fälle (10,7–42,6%) in Lyon dar. In Grenoble können 6,8% der Neuerkrankungen durch Lungenkrebs auf Luftschadstoffe zurückgeführt werden (95% Konfidenzintervall: 3,1–10,1%). Das Risiko (einer Erkrankung) war um 8 bis 20 % geringer wenn für die Abschätzung der Exposition lediglich die Daten der Hintergrundstationen verwendet wurden. Das Risiko (einer Erkrankung) war am höchsten in Wohngebieten mit mittlerer bis höherer sozialer Benachteiligung. Der höchste Anteil der Fälle, die auf eine Exposition gegenüber Luftschadstoffen zurückgeführt werden können, wurde für die Zielvariable „niedriges Geburtsgewicht" errechnet, die normalerweise in ähnlichen Studien nicht untersucht wird.

In einer Gesundheitsfolgenabschätzungs-Studie wurde die durch gegenwärtige und zukünftige Ozon-Konzentrationen bedingte Mortalitätsrate in Europa abgeschätzt. Die Ozon-Konzentrationen wurden für ein 50 km x 50 km Raster mit Hilfe des chemischen Transportmodells MATCH4 und der durch RCP4.5 abgeschätzten anthropogenen Emissionen der Vorläufersubstanzen ermittelt. Das gegenwärtige Klima für den Zeitraum 1991 - 2000 wurde mit dem zukünftigen Klima für den Zeitraum 2046–2055 verglichen. Mortalitätsdaten wurden von der WHO Datenbank „European Health for All" gewonnen. Die gerasterten Populationsdaten für Europa für die Jahre 2010 und 2050 wurden von den europäischen Projekten INTARESE und HEIMTSA übernommen. Um den wichtigsten Parameter für die zukünftige Änderung der Mortalitätsrate herauszufinden, wurde jeweils nur ein Parameter verändert, während alle anderen konstant gehalten wurden. Um die Ozon-Effekte sowohl für Kurzzeit- als auch für Langzeitexposition zu beschreiben, wurden die Koeffizienten für das relative Risiko sowohl von der europäischen APHEA2 Studie als auch von US-amerikanischen Kohortenstudien angewandt. Es wird erwartet, dass die zukünftigen Ozon-Konzentrationen ansteigen, bedingt durch Abnahme der Konzentrationen von Vorläufersubstanzen sowie Zunahme durch den Klimawandel. Dennoch wird insgesamt eine Abnahme der bodennahen Ozon-Konzentration für die Zukunft prognostiziert. Es wird geschätzt, dass gegenwärtig 44 000 vorzeitige Todesfälle in Europa auf erhöhte Ozon-Konzentration zurückzuführen sind, davon 26 000 aufgrund von Kurzzeitexposition (ausgehend von einem Cut-off Wert von 25 ppb für die Effekte). Die Ergebnisse sind am empfindlichsten bezüglich der angenommenen Cut-off Werte -

wird keine oder nur eine geringe Schwellenkonzentration angenommen, können die gesundheitlichen Auswirkungen, zumindest der Langzeitexposition, stärker ausfallen als ursprünglich abgeschätzt.

In einer anderen europaweiten Gesundheitsfolgenabschätzungs-Studie wurde der Umfang der Auswirkungen einer zukünftigen Erhöhung der Temperatur auf die Gesundheit untersucht. Die Auswirkungen wurden anhand unterschiedlicher Klimamodelle auf Basis des RCP4.5- und RCP8.5-Klimaszenarios für ein 50 km x 50 km Raster gerechnet. Generell zeigt die Abschätzung dass etwa 1% der Gesamtmortalität in den verschiedenen EU-Mitgliedstaaten auf eine erhöhte Temperatur zurückzuführen ist. Der für die Zukunft prognostizierte Umfang der Auswirkungen variiert stark in Abhängigkeit vom gewählten Modell. Die Ergebnisse für das RCP4.5-Szenario zeigen, dass 1,3 bis 4,3% der Mortalität in verschiedenen europäischen Ländern auf hohe Temperaturen zurückzuführen sind, bei Annahme eines auf neueren europäischen Studien basierenden Mittelwerts für das relative Risiko.

**Anlage IB: Summary of the final ACCEPTED report**

Changes in urban design and traffic policy, demography, climate and environmental policies, are likely to modify both outdoor and indoor air quality and therefore also public health. ACCEPTED is a research program that aims to improve our understanding of future exposure situations in cities and their impact on health, from an interdisciplinary approach. This is achieved by using various state-of-the-art atmospheric models and measurements describing effects of trends and potential changes on exposure together with epidemiological exposure-response functions.

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9. Centre Scientifique et Technique du Bâtiment (CSTB), Bernard Collignan, Emilie Powaga
10. AIRPARIF, Cecile Honoré
11. Institut scientifique de service public (ISSeP), Suzanne Remy

**Impacts of decreased emissions and climate change**

We project, using a four-model ensemble, that European anthropogenic emission decrease until 2050 will cause a strong improvement in air quality, in general having a much stronger impact than climate change on the air quality in Europe. However, this conclusion is dependent on the resolution of the emission inventory and models applied: When describing Paris at higher resolution, climate change turns out to be the most influential factor for changes in maximum ozone, whereas local emissions is the main driver for changes in particles and mean ozone. This suggests that the coarse-resolution projections could overestimate the impact of the future emission mitigation to the total ozone concentration response. Thus, the use of coarse resolution emission inventories and modeling may lead to significant biases in assessments and policy misclassification, even over areas with major regional contribution such as Stockholm. Due to climate change until the mid-century we project decreased ozone in Stockholm and particle mass and ozone in Paris, and increased particle mass in Stockholm and Brussels.

Our projections of urban scale climate change show substantial warming in the cities of Brussels and Paris, 1.6°C and 1.8°C respectively, until the 2050s horizon. We project a decrease in daytime UHI in summer connected to drying soils over rural areas and a wintertime increase in the nocturnal UHI intensity due to a decreased wind speeds. The projected increase in wintertime UHI intensity during night is projected to cause an increase in particulate matter locally in the studied urban areas. Since the increase in wintertime nocturnal UHI intensity is also where people reside and emit pollutants, it will be very important for policy makers to make sure that the reduction in anthropogenic emissions will continue to counteract this climate penalty. These results were modelled with a new urban climate downscaling technique developed within this project. Urban development and potentially more frequent extreme climate events such as heat waves are expected to cause strong impacts on the UHIs.

**Benefits of local policies**

For Stockholm we have quantified the impact of mitigation scenarios for the future. Moving traffic on a major high-way into a tunnel in Stockholm will decrease the health impacts due to reduced traffic pollution exposure in the residents, but the positive effect is cancelled due to the increased exposure among tunnel users. We further show that an increased congestion tax would have a minor effect on the exposure. Exchanging fossil fueled vehicles by electric would reduce the particulate matter exposure but it would increase the ozone exposure. However, the positive benefit of climate change on surface ozone in Stockholm is projected to be stronger. The European-wide emission reductions until 2050 causes a strong decrease in both particle mass and ozone, far exceeding the impacts of climate change or local mitigation, however, banning the use of studded tires would cause a significant decrease in $PM_{10}$, equal to about 50% of the effect of the projected European-wide emission reductions until 2050.

For evaluation of implemented control policy, we focused specifically on the impact of enforced low emission zones (LEZ) in Munich, Berlin and Augsburg and congestion charges in Stockholm. We showed that the efficiency of the LEZ depends on the strictness of the regulation as well as the size of the regulated area, and that it is most effective for health relevant components (such as soot) contained in $PM_{10}$. It means that even though the impact on $PM_{10}$ may be small, the impact on health can be stronger since the LEZ reduce exhaust particles that may have the most adverse health impacts. In order to make a proper evaluation of control policies, it is very important to monitor the impacts on traffic intensities and vehicle composition both inside and outside the zone. In addition, the evaluation of LEZ by monitoring of EC and/or OC in $PM_{10}$ would be a better strategy to demonstrate the efficacy of such measures.

**Outdoor and indoor exposure**

Indoor exposure accounts for 90% and 87% of total population exposure to outdoor near-surface ozone and $PM_{2.5}$ according to a model for Greater Paris, and the exposure to ozone is generally higher for the residents of the rural area. Because the buildings in downtown Paris are older than over the rural area, leading to a higher infiltration rate of ozone, the urban increment of outdoor concentrations has relatively small effect on personal exposures. Retired people, being one of the vulnerable groups of the rural area, living in old houses, are exposed to 40% higher concentrations than the population average exposure.

Increasing building insulation for saving energy will lead to lower exposure to outdoor near-surface ozone, whereas the impact on $PM_{2.5}$ is weak.

**Health effects in vulnerable groups**

One large study was done with data from Greater Stockholm to examine possible associations between the traffic pollution situation at the home address and preterm delivery and pregnancy-induced hypertensive disorders. There was an association between elevated traffic pollution exposure during pregnancy and pregnancy-induced hypertensive disorders. A 10 $\mu g/m^3$ increase in the pregnancy average NOx level at the home address resulted in an increased risk of approximately 17% (95% CI 10-26). There was also a tendency of a higher risk of preterm delivery in relation to higher levels of NOx. A second phase of the study investigated the association with traffic-related particles, and found similar associations, with an increased risk of pre-eclampsia, gestational hypertension and risk of preterm delivery.

Moreover, in another study from Belgium the exposure to particulate matter during pregnancy was associated with alteration in the cord blood levels of metabolites indicating an altered inflammatory state of the newborn at birth induced by air pollution during in utero life.

The triggering effect of extreme temperatures on the risk of preterm birth was investigated in Flanders (Belgium) and in Stockholm (Sweden). Considering the cumulative effect of temperature up to 6 days before delivery (lag 0–6) in Flanders, the risk of preterm birth increased by 8.8% (95% CI 0.7%-17.5%) for an increase in minimum temperature from 8.3°C (50th percentile) to 16.3°C (95th percentile). The corresponding estimate for an increase in maximum temperature from 14.7°C (50th percentile) to 26.5°C (95th percentile) was 9.1% (95% CI -0.3%-19.2%). In Stockholm, the risk of preterm birth during summer increased by 4%–5% when the mean temperature reached the 75th percentile four weeks earlier, with inconsistent associations for more extreme temperatures. In both studies, no clear evidence for an effect of cold on preterm birth was found.

**Estimation of local and European wide health impacts**

Risk assessment studies often ignore within-city spatial variations of air pollutants. Our objective was to quantify the attributable risk of fine particulate matter ($PM_{2.5}$) in 2 urban areas using fine-scale air pollution modeling and characterizing how this impact varied according to social deprivation. In Grenoble and Lyon areas (0.4 and 1.2 million inhabitants, respectively) in 2012, $PM_{2.5}$ exposure was estimated on a 10x10 m grid by coupling a dispersion model to population density. Outcomes were mortality, lung cancer and term low birth weight incidences. The numbers of outcome cases attributable to air pollution were estimated overall and stratifying areas according to the European Deprivation Index (EDI), by comparing the actual situation to the World Health Organization guideline value (10 $\mu g/m^3$ yearly average) as reference (counterfactual) level. Estimations were repeated assuming spatial homogeneity of air pollutants within urban area. Median $PM_{2.5}$ levels were 18.1 and 19.6 $\mu g/m^3$ in Grenoble and Lyon urban areas, respectively, corresponding to 114 (5.1% of total, 95% confidence interval, CI, 3.2–7.0%) and 491 non-accidental deaths (6.0% of total, 95% CI 3.7–8.3%) attributable to long-term exposure to $PM_{2.5}$, respectively. Attributable term low birth weight cases represented 23.6% of total cases (9.0–37.1%) in Grenoble and 27.6% of cases (10.7–42.6%) in Lyon. In Grenoble, 6.8% of incident lung cancer cases were attributable to air pollution (95% CI 3.1–10.1%). Risk was lower by 8 to 20% when estimating exposure through background stations. Risk was highest in neighborhoods with intermediate to higher social deprivation. The highest proportion of cases attributed to air pollution exposure was in this case estimated for term low birth weight, an outcome usually not included in similar assessments.

In one health impact assessment current and future projected ozone induced mortality in Europe is estimated. European near surface ozone concentrations were modelled at a grid size of 50x50 km using the chemistry-transport model MATCH4 with anthropogenic precursor emissions from RCP4.5. The current climate as 1991–2000 was compared to the future climate as 2046–2055. Data on mortality was attained from WHO's European Health for All Database. Gridded population data for Europe in 2010 and 2050 was taken from the European projects INTARESE and HEIMTSA. To see main driver of future mortality change, only one of the drivers were modified and other factors kept constant in each step. In order to describe the ozone effects due to short-term and long-term exposures, the relative risk coefficients from European APHEA2 and American cohort studies are applied. The future ozone levels are expected to increase due to decrease in ozone precursors and increase due to climate change. However, taken together the near ground ozone levels are expected to be smaller in the future. Exposure to near-surface background ozone in Europe is currently expected to cause 44,000 premature deaths in Europe annually, from which 26,000 due to short-term exposure (assuming a cut-off for effects at 25 ppb). The results are most sensitive to implemented cut-off levels, if there is no or a low threshold, at least for long-term exposure, the true health impacts could be bigger than previously assessed.

In another European wide health impact assessment the range of future impacts from high temperatures was studied using 8 different climate models from the RCP45 and RCP85 climate scenarios on a 50x50km grid over Europe. The general pattern of the estimations is that around 1% of total mortality in the different countries is results of elevated temperatures. For the future period the range of impacts from the different models shows large variation. The results for RCP45 shows an estimated annual mortality related to heat in the different countries ranging from 1.3 to 4.3% when using a mean of relative risks from a recent European study.

**Anlage II:**

**ACCEPTED Final report**

**(Dateiname: ACCEPTED Final Report.pdf)**

## Anlage III:

**ACCEPTED Report on Low Emission and Congestion Charge Zones in Europe with impact assessment studies for Augsburg, Munich, Berlin and Stockholm**

**(Dateiname: LEZ_report_2014_June19.pdf)**

# ACCEPTED

*Final report*

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# 2. SUMMARY

## Publishable summary

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11. Institut scientifique de service public (ISSeP), Suzanne Remy

### Impacts of decreased emissions and climate change

We project, using a four-model ensemble, that European anthropogenic emission decrease until 2050 will cause a strong improvement in air quality, in general having a much stronger impact than climate change on the air quality in Europe. However, this conclusion is dependent on the resolution of the emission inventory and models applied: When describing Paris at higher resolution, climate change turns out to be the most influential factor for changes in maximum ozone, whereas local emissions is the main driver for changes in particles and mean ozone. This suggests that the coarse-resolution projections could overestimate the impact of the future emission mitigation to the total ozone concentration response. Thus, the use of coarse resolution emission inventories and modeling may lead to significant biases in assessments and policy misclassification, even over areas with major regional contribution such as Stockholm. Due to climate change until the mid-century we project decreased ozone in Stockholm and particle mass and ozone in Paris, and increased particle mass in Stockholm and Brussels.

Our projections of urban scale climate change show substantial warming in the cities of Brussels and Paris, 1.6°C and 1.8°C respectively, until the 2050s horizon. We project a decrease in daytime UHI in summer connected to drying soils over rural areas and a

wintertime increase in the nocturnal UHI intensity due to a decreased wind speeds. The projected increase in wintertime UHI intensity during night is projected to cause an increase in particulate matter locally in the studied urban areas. Since the increase in wintertime nocturnal UHI intensity is also where people reside and emit pollutants, it will be very important for policy makers to make sure that the reduction in anthropogenic emissions will continue to counteract this climate penalty. These results were modelled with a new urban climate downscaling technique developed within this project. Urban development and potentially more frequent extreme climate events such as heat waves are expected to cause strong impacts on the UHIs.

## Benefits of local policies

For Stockholm we have quantified the impact of mitigation scenarios for the future. Moving traffic on a major high-way into a tunnel in Stockholm will decrease the health impacts due to reduced traffic pollution exposure in the residents, but the positive effect is cancelled due to the increased exposure among tunnel users. We further show that an increased congestion tax would have a minor effect on the exposure. Exchanging fossil fueled vehicles by electric would reduce the particulate matter exposure but it would increase the ozone exposure. However, the positive benefit of climate change on surface ozone in Stockholm is projected to be stronger. The European-wide emission reductions until 2050 causes a strong decrease in both particle mass and ozone, far exceeding the impacts of climate change or local mitigation, however, banning the use of studded tires would cause a significant decrease in PM10, equal to about 50% of the effect of the projected European-wide emission reductions until 2050.

For evaluation of implemented control policy, we focused specifically on the impact of enforced low emission zones (LEZ) in Munich, Berlin and Augsburg and congestion charges in Stockholm. We showed that the efficiency of the LEZ depends on the strictness of the regulation as well as the size of the regulated area, and that it is most effective for health relevant components (such as soot) contained in $PM_{10}$. It means that even though the impact on $PM_{10}$ may be small, the impact on health can be stronger since the LEZ reduce exhaust particles that may have the most adverse health impacts. In order to make a proper evaluation of control policies, it is very important to monitor the impacts on traffic intensities and vehicle composition both inside and outside the zone. In addition, the evaluation of LEZ by monitoring of EC and/or OC in $PM_{10}$ would be a better strategy to demonstrate the efficacy of such measures.

## Outdoor and indoor exposure

Indoor exposure accounts for 90% and 87% of total population exposure to outdoor near-surface ozone and PM2.5 according to a model for Greater Paris, and the exposure to ozone is generally higher for the residents of the rural area. Because the buildings in downtown Paris are older than over the rural area, leading to a higher infiltration rate of ozone, the urban increment of outdoor concentrations has relatively small effect on personal exposures. Retired people, being one of the vulnerable groups of the rural area, living in old houses, are exposed to 40% higher concentrations than the population average exposure.

Increasing building insulation for saving energy will lead to lower exposure to outdoor near-surface ozone, whereas the impact on PM2.5 is weak.

## Health effects in vulnerable groups

One large study was done with data from Greater Stockholm to examine possible associations between the traffic pollution situation at the home address and preterm delivery and pregnancy-induced hypertensive disorders. There was an association between elevated traffic pollution exposure during pregnancy and pregnancy-induced hypertensive disorders. A 10 µg/m3 increase in the pregnancy average NOx level at the home address resulted in an increased risk of approximately 17% (95% CI 10-26). There was also a tendency of a higher risk of preterm delivery in relation to higher levels of NOx. A second phase of the study investigated the association with traffic-related particles, and found similar associations, with an increased risk of pre-eclampsia, gestational hypertension and risk of preterm delivery.

Moreover, in another study from Belgium the exposure to particulate matter during pregnancy was associated with alteration in the cord blood levels of metabolites indicating an altered inflammatory state of the newborn at birth induced by air pollution during in utero life.

The triggering effect of extreme temperatures on the risk of preterm birth was investigated in Flanders (Belgium) and in Stockholm (Sweden). Considering the cumulative effect of temperature up to 6 days before delivery (lag 0–6) in Flanders, the risk of preterm birth increased by 8.8% (95% CI 0.7%-17.5%) for an increase in minimum temperature from 8.3°C (50th percentile) to 16.3°C (95th percentile). The corresponding estimate for an increase in maximum temperature from 14.7°C (50th percentile) to 26.5°C (95th percentile) was 9.1% (95% CI -0.3%-19.2%). In Stockholm, the risk of preterm birth during summer increased by 4%–5% when the mean temperature reached the 75th percentile four weeks earlier, with inconsistent associations for more extreme temperatures. In both studies, no clear evidence for an effect of cold on preterm birth was found.

### Estimation of local and European wide health impacts

Risk assessment studies often ignore within-city spatial variations of air pollutants. Our objective was to quantify the attributable risk of fine particulate matter (PM2.5) in 2 urban areas using fine-scale air pollution modeling and characterizing how this impact varied according to social deprivation. In Grenoble and Lyon areas (0.4 and 1.2 million inhabitants, respectively) in 2012, PM2.5 exposure was estimated on a 10x10 m grid by coupling a dispersion model to population density. Outcomes were mortality, lung cancer and term low birth weight incidences. The numbers of outcome cases attributable to air pollution were estimated overall and stratifying areas according to the European Deprivation Index (EDI), by comparing the actual situation to the World Health Organization guideline value (10 µg/m3 yearly average) as reference (counterfactual) level. Estimations were repeated assuming spatial homogeneity of air pollutants within urban area. Median PM2.5 levels were 18.1 and 19.6 µg/m3 in Grenoble and Lyon urban areas, respectively, corresponding to 114 (5.1% of total, 95% confidence interval, CI, 3.2–7.0%) and 491 non-accidental deaths (6.0% of total, 95% CI 3.7–8.3%) attributable to long-term exposure to PM2.5, respectively. Attributable term low birth weight cases represented 23.6% of total cases (9.0–37.1%) in Grenoble and 27.6% of cases (10.7–42.6%) in Lyon. In Grenoble, 6.8% of incident lung cancer cases were attributable to air pollution (95% CI 3.1–10.1%). Risk was lower by 8 to 20% when estimating exposure through background stations. Risk was highest in neighborhoods with intermediate to higher social deprivation. The highest proportion of cases attributed to air pollution exposure was in this case estimated for term low birth weight, an outcome usually not included in similar assessments.

In one health impact assessment current and future projected ozone induced mortality in Europe is estimated. European near surface ozone concentrations were modelled at a grid size of 50x50 km using the chemistry-transport model MATCH4 with anthropogenic precursor emissions from RCP4.5. The current climate as 1991–2000 was compared to the future climate as 2046–2055. Data on mortality was attained from WHO's European Health for All Database. Gridded population data for Europe in 2010 and 2050 was taken from the European projects INTARESE and HEIMTSA. To see main driver of future mortality change, only one of the drivers were modified and other factors kept constant in each step. In order to describe the ozone effects due to short-term and long-term exposures, the relative risk coefficients from European APHEA2 and American cohort studies are applied. The future ozone levels are expected to increase due to decrease in ozone precursors and increase due to climate change. However, taken together the near ground ozone levels are expected to be smaller in the future. Exposure to near-surface background ozone in Europe is currently expected to cause 44,000 premature deaths in Europe annually, from which 26,000 due to short-term exposure (assuming a cut-off for effects at 25 ppb). The results are most sensitive to implemented cut-off levels, if there is no or a low threshold, at least for long-term exposure, the true health impacts could be bigger than previously assessed.

In another European wide health impact assessment the range of future impacts from high temperatures was studied using 8 different climate models from the RCP45 and RCP85 climate scenarios on a 50x50km grid over Europe. The general pattern of the estimations is that around 1% of total mortality in the different countries is results of elevated temperatures. For the future period the range of impacts from the different models shows large variation. The results for RCP45 shows an estimated annual mortality related to heat in the different countries ranging from 1.3 to 4.3% when using a mean of relative risks from a recent European study.

## Project summary

These are the key findings by the ACCEPTED project:

- It is important to describe urban areas at a high resolution when climate and air pollution is modeled. The use of coarse resolution emission inventories and modeling may lead to significant biases in assessments and policy misclassification, even over areas with major regional contribution.
- Using high-resolution climate modelling, substantially warmer cities by the 2050s are projected, with changes also in the urban heat island (UHI) intensity.
- The use of fine-resolution air pollution model also led to an increased estimated impact of air pollution in two urban areas, compared to the commonly used approach relying on background monitoring stations.
- The introduction of low emission zones and congestion charges systems can reduce traffic-related pollutants significantly, soot or elemental carbon are more relevant indicators than PM2.5 or PM10.
- In a city like Stockholm, change in European emissions has the largest impact on future concentrations of particle mass and near-surface ozone. However, removing the studded tires in Stockholm 2030 would also have a strong benefit on annual mean PM10.
- People are exposed to a large proportionction of the outdoor pollutants during time spent indoors, especially in old houses with higher infiltration rates.
- Pregnant women and their unborn children seem to be vulnerable with respect to traffic-related air pollution and extreme heat, however outcomes such as preterm birth or low birth weight are usually not included in the health impact assessments.
- In the cities of Grenoble and Lyon, between 3 and 8% of non-accidental deaths and new lung cancer cases were attributable to exposure to fine particulate matter. Particulate matter may be responsible for an even larger proportion of term low birth weight cases.

# 3. OBJECTIVES OF THE RESEARCH

The underlying hypothesis of this project is that air pollution exposure and urban climate interacts and significantly affects the health of humans. Interventions and policies may, however, reduce the health impacts, and such benefits can now be better estimated than before. The project has, in particular, addressed questions and limitations important for such assessments.

In work package 1 (WP1) we develop and apply existing models that predict changed urban air quality and climate. The models are used to assess future climate and future levels of air pollution in different European cities following alternative mitigation strategies to improve the urban environment and global climate change. We here evaluate the impact of the implementation of Low Emission Zones (LEZs) in several cities.

The calculated concentrations of urban air pollutants ($O_3$, PM etc.) are used in WP2 together with data on building structure and relationships between indoor and outdoor air quality, to determine the changes in exposure to air pollutants in investigated emission scenarios. Climate and air quality data are also used for the exposure assessments that is developed into health impact assessments in WP4, partly relying on novel exposure-response functions for vulnerable groups. Inclusion of effects on vulnerable groups, such as foetuses and young children would significantly increase the value of health impact assessments and increase the motivation of mitigation policies to reduce air pollution in cities. Such exposure-response relations are developed in WP3.

In WP4, scenario-based health impact assessments (HIAs) we combine exposure information from climate models, emission scenarios, policy evaluation studies and concentration calculations with exposure-response functions (ERFs) from new or previously published epidemiological studies. We study how the estimated health benefits from air pollution control policies in cities are sensitive to exposure data and modelling, inclusion of effects on vulnerable groups and on demographic trends. For the study areas, we use population data in grids, and extract baseline rates from regional and national registers.

# 4. PROJECT ACTIVITIES AND ACHIEVEMENTS

The scientific work has been divided into four work packages, where WP1 dealt with air quality in cities and WP2 dealt with the relation between indoor and outdoor air quality, activity patterns and personal exposure modelling. WP3 included the epidemiological studies of vulnerable groups and studies of exposure response functions. The Health Impact Assessments were be performed in WP4, by using information collected in the other WPs.

Each WP had a leader, and the project was overviewed by a steering group and a coordinator. There has been a number of workshops and meetings to plan and discuss the work and conclusions:

- Kick-off meeting in Stockholm, 28th of February – 1st of March, 2013
- Workshop, WP1 in Basel, 19th of August, 2013
- Workshop, WP2 in Paris, 22nd of November, 2013
- Workshop, WP3 in Brussels, 4th of March, 2014
- LEZ Seminar in Stockholm, 17th – 18th of June, 2014
- Workshop, WP4 in Augsburg, 11th – 12th of September, 2014
- Final meeting in Brussels, 22nd of October, 2015

## Factual description, specifying the input of each participant

### Partner 1: Umeå University (coordinator), Bertil Forsberg

Umeå University (UmU) has been coordinating the Accepted Project with Bertil Forsberg as the coordinator and Kristina Lindblom as the project administrator. Other participants from UmU are: Christofer Åström, Daniel Oudin, Hans Orru, David Olsson, Ana Vicedo, Kristie Ebi and Johan Sommar. The coordinating centre has organised the project meetings including the final meeting, steering group meetings and meeting minutes (mostly conference calls). The coordinator and steering group has followed the work in the different work packages, and coordinated the mid-term report (Deliverable 2) and final project report. UmU has also in lines with the communication strategy been responsible for the project website, the Accepted newsletter and the mailing list. As coordinating centre UmU is also responsible for the Report on guidelines for health impact assessment of future climate and air pollution scenarios (Deliverable 9) and the editing of a Project summary report for end-users (Deliverable 10).

UmU has performed research mainly related to WP2-4. In WP2 the I/O ratio for ozone in thirty-four Swedish (City of Uppsala) residences was investigated in relation to type of building and ventilation. In WP3 epidemiological Swedish studies of temperature, air pollution and birth outcome (Deliverable 6a, 6c) and studies of studies of air pollution and childhood asthma has been undertaken (Deliverable 7), and some work in collaboration with Hasselt University. UmU also organised a WP3 workshop. In WP4 UmU has organised a workshop on Health impact assessment in Augsburg, and been involved in a number of case studies in terms of health impact assessments of traffic policy and impacts of climate change (see papers and manuscripts in the publication list).

UmU has been represented at meetings and workshops organized within the Accepted Project.

## Partner 2: Swedish Meteorological and Hydrological Institute (SMHI), Camilla Andersson

Camilla Andersson has been the main responsible for the SMHI participation/contribution. She was also the leader of WP1 and a member of the steering committee. Other participants from SMHI are: Lars Gidhagen, Cecilia Bennet and Magnuz Engardt.

SMHI (Camilla Andersson, Lars Gidhagen and Cecilia Bennet) have evaluated the low emission zone (LEZ) in Augsburg, Germany, using a dispersion model in the AIRVIRO system. The activities were planned during a WP1 seminar in Sept 2013 (Milestone 2, Deliverable 1). The analyses (Milestone 7) were made on time. The results of these analyses were published in a report (Deliverable 3). The LEZs have given effect in other cities studied by other partners (Stockholm, Berlin and Munich), but in Augsburg the results are less clear which may be due to a too small control zone.

For Stockholm SMHI (Camilla Andersson and Magnuz Engardt) conducted modelling of future climate and emission scenarios (Milestone 11). These scenarios were planned at a work shop in Nov 2013 (Milestone 2, Deliverable 1). The results are published in a report (Deliverable 4) on the effect of future climate and control policies on the air quality in European cities. Scientific papers in pipeline/compilation include Markakis et al. (2015 accepted in ACPD) comparing the Stockholm and Paris scenarios, Lacressoniere et al., (2015; 2015 manuscript) and Watson et al. (2015; 2015 manuscript) on the present and future European air quality and the results are also input to health impact assessments for Europe and Stockholm with four manuscripts based on the results (Andersson et al., 2015; Oudin D. et al., 2015; Orru et al., 2015; Åström C. et al., 2015).

The SMHI (Camilla Andersson) contribution also includes: A mid-term report on policy relevance and knowledge transfer which was completed on time (Deliverable 2) and the final results were presented at the final project seminar in Brussels (milestone no 6). SMHI attended all project meetings (Milestones 1-6). The WP1 work shop was split into two workshops. The first work shop on LEZ in Basel was arranged by SMHI (Milestone 2), the second was a joint meeting between WP1 and 2 in Paris (Milestones 2-3) on future modelling scenarios. SMHI helped arrange some of the project meetings (Milestones 2, 3, 5, 6) and participated as a member of the steering committee and WP leader of WP1. Camilla was the main author responsible for the report on future scenarios (Delivery 4).

## Partner 3: City of Stockholm, Christer Johansson

Christer Johansson has been the main responsible for Stockholm's contribution. The work has included:

- Organisation of international conference in Stockholm on Low Emission Zones, June 17-18, 2014.
- Main responsibility for writing ACCEPTED project report on European Low Emission Zones (Delivarable 3).
- Calculations of air quality, exposure and health impacts of different mitigation action scenarios in Stockholm including congestion tax, low emission zone, fossil fuel free fleet and electric vehicle fleet (Deliverables 6-7, Papers and manuscripts in publication list). Contributing to estimating climate change effects on PM, ozone and NOx.
- Co-authoring and contribution to the ACCEPTED project report on the effect of future climate and control policies on air quality in Stockholm, Paris and Brussels (Deliverable 4).

Christer Johansson and Sanna Silvergren participated in the workshop on Low emission zone in Basel (August 19th 2013).

Participation in ACCEPTED Workshop on Health Impact Assessment, The Environmental Science Center (WZU), University of Augsburg, September 11-12 2014.

Participation in the final conference of ACCEPTED October 22-23, 2015, Brussels.

### Partner 4: Hasselt University, Tim Nawrot

Tim Nawrot, Hasselt University, has been work-package leader for WP3 containing the epidemiological studies of vulnerable groups included in the Accepted Project (Deliverables 6-8). Other participants from Hasselt University have been Bianca Cox and Dries Martens. Hasselt University assisted UmU in arranging the final meeting in Brussels in October 2015.

Hasselt University has studied air pollution and early life metabolomic changes by quantification of cord blood plasma oxylipin profiles in early life, as targets of the pregnancy exposome, in association with in utero exposure to particulate matter (PM) air pollution. 197 cord blood plasma samples form the ENVIRONAGE birth cohort were included. The indoor air pollution levels of households of pregnant women were quantified by partner 11 ISSeP. The risk of preterm birth in association with extreme temperatures in Belgium was studied using data on 446,110 singleton natural deliveries between 1998 and 2011.

For papers and manuscripts see the list of publications.

### Partner 5: Royal Meteorological Institute of Belgium (RMI), Rafiq Hamdy

The Royal Meteorological Institute of Belgium has performed research mainly on WP1 dealing with future climate change over urban areas and the future air quality. During the project a new high-resolution urban dynamical downscaling method has been developed and successfully applied over the city of Brussels and Paris. Also the output of these climate simulations has been coupled to a Chemical Transport Model, CHIMERE in order to investigate the future air quality (see publication list). Participants from RMI has presented results from Accepted at several scientific conferences and also communicated these results to a broader audience (see below).

### Partner 6: University of Augsburg, Josef Cyrus

WZU has contributed to the evaluation of the Low Emission Zone (LEZ) effects in three German cities: Augsburg, Berlin and Munich using monitoring data and dispersion modelling. The dispersion modeling was done by SMHI using a model in the AIRVIRO system and by WZU using the IMMIS model. For both approaches WZU collected and provided the emission inventory, traffic counting, monitoring and meteorological data.

In addition to the modeling, the LEZs in Berlin, Munich and Augsburg were evaluated by statistical analysis of measured PM10 data. The effectiveness of LEZs was evaluated at different types of monitoring sites (traffic site and urban background site). The results of these analyses were published in a study report (Deliverable 3) as well as national and international journals. The main result is that the LEZs have led to lowering of fine particles levels in Berlin and Munich, but in Augsburg the results are less clear which may be due to a too small area of the LEZ.

WZU analyzed the results of the personal measurements for PNC, PM2.5 and BS in different scenarios and handed it over to CSTB for model validation.

Josef Cyrys and Jianwei Gu participated in the first WP1 workshop on LEZs in Basel (August 19th 2013). Josef Cyrys participated in the second joint workshop of WP1 and WP2 on future modelling scenarios in Paris (November 21-22 2013).

Josef Cyrys participated in the international conference in Stockholm on Low Emission Zones, June 17-18, 2014.

WZU organized the ACCEPTED workshop on Health Impact Assessment in Augsburg (September 11-12 2014).

Josef Cyrys participated in the final ACCEPTED conference in Brussels (October 22-23, 2015).

### Partner 7: Centre National de la Recherche Scientifique (CNRS), Myrto Valari

Myrto Valari has been the deputy coordinator and leader of WP2. Under the framework of the ACCEPTED project CNRS has developed a personal exposure model for the greater Paris region. The model integrates indoor and outdoor air-quality with time activity diaries including population mobility patterns. To develop the population's personal diaries we used input from the local environmental agency (AIRPARIF). In addition the "Centre scientifique et technique du batiment" (CSTB) has provided a database of indoor/outdoor infiltration rates using the air-renewal model SIREN. These rates were used to determine exposure indoors. The outdoor air-quality was simulated with the chemical transport model CHIMERE. Boundary conditions for the local-scale simulations were provided by the Laboratoire des Sciences du Climat et de l'Environnement (LSCE) under the IMPACT2C FP7 project. The exposure model was used to simulate present-time and mid-21$^{st}$ century population exposure under climate change and building stock scenarios.

### Partner 8: Institut national de la santé et de la recherche médicale (INSERM), Rémy Slama

Inserm conducted a health assessment of the impact of exposure to fine particulate matter in Grenoble and Lyon urban areas, in collaboration with Air Rhône-Alpes (the local air quality monitoring network) and ACCEPTED partners. This health impact assessment study had several originalities, namely 1) reliance on a fine-scale dispersion model, and comparison of this approach with reliance on monitoring stations, as is generally the case in former similar studies; 2) consideration of term low birth weight as a relevant outcome in health impact assessment studies; 3) study of possible heterogeneity between social categories in the health impact of fine particulate matter. In addition, the expected benefic of several scenarios of reduction of air pollution has been evaluated. This work resulted in scientific communications and also in the successful defense of a PhD, by Xavier Morelli (January 2016), which included results from ACCEPTED project to a large extent.

Inserm participated to the ACCEPTED WP1 workshop in Paris (21 November 2013).

Inserm participated to the ACCEPTED workshop on Health Impact Assessment in Augsburg (September 11-12 2014).

Inserm participated in the final ACCEPTED conference in Brussels (October 22-23, 2015).

Inserm engaged in discussions with the local authorities regarding the implementations of measures aiming at reducing air pollution, such as those existing in other areas covered by ACCEPTED project, and presented the results of the health impact assessment to the Mayor of Grenoble.

### Partner 9: Centre Scientifique et Technique du Bâtiment (CSTB), Bernard Collignan, Emilie Powaga

CSTB were mostly involved in WP2 dedicated to determine changes in exposure to air pollutants in the investigated climate and emission scenarios, using the results from changes urban air quality and climate (WP1). This has been achieved by using an integrated view accounting both for indoor and outdoor air pollution as well as for population time-activity. Numerical calculations were made using ventilation model developed at CSTB to assess outdoor ozone and fine particle ($PM_{2.5}$) transfer indoors as a function of building characteristics, meteorological conditions and outdoor air-pollution. The indoor-outdoor (I/O) ratios were sent to LMD/CNRS partner to conduct the exposure modelling.

The work carried out by CSTB during the project can be listed as:

- An extensive state of the art was conducted to identify the factors that may influence the indoor to outdoor concentration ratios of two specific outdoor pollutants related with road traffic and climate change: $PM_{2.5}$ and Ozone.

- A description of the current French building stock and its evolution with regard to usage, thermal and ventilation regulations was made in order to define building typologies that can be used in calculations.
- An adaptation of the ventilation model SIREN was made to determine relationship between indoor and outdoor for Ozone and PM2.5 and gave the ability to assess the indoor concentration of this pollutant through time depending on outdoor scenarios (provided by LMD/CNRS partner) and building typologies. An experimental confrontation of the numerical model was made and gave satisfactory results.
- Some sensitivity studies were conducted using the numerical model to determine which parameters have an influence on outdoor / indoor pollutants transfer. It has been determined that only the parameter of building age of construction (related to air-tightness and ventilation principle and systems) have an impact on the transfer of pollutant considered into the building (Ozone and PM2.5).
- In agreement with LMD/CNRS partner, the final epidemiologic study focus on the Ile-de-France region during two periods of several years each: one present-time period centred around 2010 and one future-time period centred around 2050. Some buildings typologies across age of construction and usage (dwellings, offices and schools) were defined to represent qualitatively the Ile-de-France building stock.
- Finally, it was observed that outdoor pollutant variation across time period have more impact on I/O ratios than building configuration.

### Partner 10: AIRPARIF, Cecile Honoré

AIRPARIF was involved in WP1 and WP2 in the framework of the ACCEPTED project.

This section presents the work carried out by Airparif during the project and the main outcomes:

- WP1: AIRPARIF had to build several prospective emissions inventories for Paris and the Ile-de-France region for NOx, PM10, PM2.5 and NMVOC and for different horizon of time: 2020, 2030 and 2050. These scenarios have taken into account regional and national local policies for road traffic, residential and industry sectors. These emissions results had been injected as inputs in Chimere model run by LMD.
- WP2: AIRPARIF provided LMD data of Ile-de-France such as outdoor concentrations or transport concentrations and gave assistance and expertise to build the exposure model built by LMD.

### Partner 11: Institut scientifique de service public (ISSeP), Suzanne Remy

Under the ACCEPTED project, the main task of the ISSeP was to perform indoor measurements of VOC and NO2 in 200 houses of the Belgian ENVIRONAGE birth cohort during the second month of the life of new-borns. Pollutants were selected based on their toxicity and their ubiquity in indoor air. The sampling campaign occurred from January 2014 to august 2015. Passive sampling was used for this indoor air monitoring. The sampling tubes were prepared and provided by the ISSeP but the sampling campaign in the houses was organized by U-Hasselt. After a two week sampling, the analysis were done in the laboratories of the ISSeP (the analysis of the BTEX were done by M. Gohy, the aldehydes analysis were done by V. Massen and the NO2 analysis was made by A. Minet).

A short questionnaire was developed to apprehend the relations between indoor air pollutants levels and their potential determinants. Questions focused on housing, ventilation behaviour, renovation activities and indoor emissions sources. Regarding renovation activities, questions concerned the last 6 months prior to the sampling. The questions were asked to the parents by a scientist (U-Hasselt) during the sampling time. Most of the answers were of binary type (yes or no). This work was achieved by the end of October 2015.

Descriptive statistics were computed for each VOC and for NO2 and compared to the concentrations found in Europe or other industrialized country. To evaluate the indoor air quality a health risk assessment was conducted based on Toxicological reference value for lifetime inhalation exposure for the measured pollutants. Exploratory analyses were performed to identify factors that might influence VOC and NO2 levels, focusing on 14 indoor residence characteristics and activities selected from the questionnaire. Relations between pollutants concentrations, behaviour and indoor emissions sources were highlighted with the nonparametric Wilcoxon-Mann-Whitney statistical test (U-Hasselt).

To study the relationships between the NO2 concentrations within and outside the houses, indoor measures were plotted against the distance between houses and roads (U-Hasselt) but no association was observed. Indoor air measures will be plotted with interpolated air quality data when they will be available.

### Results

A dataset of personal exposure to indoor VOCs and NO2 is available for 212 neonates of the Flemish ENVIRONNAGE birth cohort. This dataset will allow investigating associations with biomarkers that might operate as early warnings in vulnerable population groups. Furthermore the results will help establishing relevant dose-response associations that can be used for policy evaluations of the current EU thresholds.

BTEX, aldehydes and the NO2 were detected at measurable concentrations in every house.

Large differences in measured concentrations were found between the houses indicating that exposure to indoor pollution is unequal and that individual exposure data are particularly relevant when dealing with indoor air and health impacts. In few houses (<5%) concentrations were very high and were therefore far above guidelines for health protection.

The median concentrations found in the houses in this study followed the same trend than those found in other studies made in Europe, Australia and North America between 2003 and 2015.

The concentrations found in our study are usually even lower than in all other studies. This might be related to the decision of the Flemish government that came in force in 2004, which aims at promoting indoor environment quality.

Individual cancer risk exceeded the upper level of the interval of acceptability in 75% of the houses. As new-borns spent most of their time at home, they are also the most exposed to these carcinogenic.

Although 50 % of houses are located in urban or suburban areas, no relationship was found between the indoor levels and the house environment or the distance to a road.

The answers to the questionnaire revealed that most of the parents have integrated the necessity to aerate the house by opening windows every day. On the other hand the period surrounding the birth is an intensive renovation period with subsequently VOCs emissions (50% of the parents undertook renovations works and 30% bought new furniture for the baby's room).

## Scientific outcomes

The project has led to a number of scientific publications, reports, abstracts and presentations at scientific and stake holder meetings. In this section a description follows of a selection of the scientific outcomes of the project, more are to be found in the list of scientific publications.

### Impacts of decreased emissions and climate change

We have developed and applied models to describe future change in European and urban scale climate and air quality. The projected European anthropogenic emission decrease until 2050 will cause a strong decrease in air pollution, in general having a much stronger impact than climate change on the air quality in Europe. This has been shown previously and is supported by this work as well, where we use a four-model coarse-scale ensemble covering Europe to study the impact of climate and emission change until mid-century (Watson et al, 2015; Lacressoniere et al, 2015).

We have shown that these conclusions are dependent on the resolution (Markakis et al., 2015): When describing Paris at 4km resolution, climate change turns out to be the most influential factor for changes in maximum ozone, whereas local emissions is the main driver for changes in particles and mean ozone. This suggests that the coarse-resolution projections could overestimate the impact of the future emission mitigation to the total ozone concentration response. Thus, it is important to describe urban areas at a high resolution. The use of coarse resolution emission inventories and modeling may lead to significant biases in assessments and policy misclassification, even over areas with major regional contribution such as Stockholm.

Due to climate change until the mid-century we project decreased ozone in Stockholm and particle mass and ozone in Paris, and increased particle mass in Stockholm and Brussels (Andersson et al, 2015; Markakis et al, 2015).

Our simulations of urban scale climate change project substantial warming in the cities of Brussels and Paris, 1.6ºC and 1.8ºC respectively, until the 2050s horizon (Hamdi et al., 2014; 2015). We project a decrease in daytime UHI in summer connected to drying soils over rural areas and a wintertime increase in the nocturnal UHI intensity due to a decreased wind speeds.

The projected increase in wintertime UHI intensity during night is projected to cause an increase in particulate matter locally in the studied urban areas. Since the increase in wintertime nocturnal UHI intensity is also where people reside and emit pollutants, it will be very important for policy makers to make sure that the reduction in anthropogenic emissions will continue to counteract this climate penalty.

These results were modelled with a new urban climate downscaling technique developed within this project (Hamdi et al, 2014). Urban development and potentially more frequent extreme climate events such as heat waves are expected to cause strong impacts on the UHIs.

### Impacts of local policies on exposure

For Stockholm we have quantified the impact of mitigation scenarios for the future (Andersson et al, 2015). Moving traffic on a major high-way into a tunnel in Stockholm will decrease traffic pollution exposure in the residents, but the positive effect is cancelled due to the increased exposure among tunnel users (Orru et al, 2015).

We further show that an increased congestion tax would have a minor effect on the exposure. Exchanging fossil fuelled vehicles by electric would reduce the particulate matter exposure but it would increase the ozone exposure. However, the positive benefit of climate change on surface ozone in Stockholm is projected to be stronger.

The European-wide emission reductions until 2050 causes a strong decrease in both particle mass and ozone, far exceeding the impacts of climate change or local mitigation, however, banning the use of studded tires would cause a significant decrease in PM10, equal to about 50% of the effect of the projected European-wide emission reductions until 2050.

For evaluation of implemented control policy, we focused specifically on the impact of enforced low emission zones (LEZ) in Munich, Berlin and Augsburg and congestion charges in Stockholm (Johansson et al, 2014; Cyrys et al, 2014; Fernsterer et al, 2014). We showed that the efficiency of the LEZ depends on the strictness of the regulation as well as the size of the regulated area, and that it is most effective for health relevant components (such as soot) contained in $PM_{10}$. It means that even though the impact on $PM_{10}$ may be small, the impact on health can be stronger since the LEZ reduce exhaust particles that may have the most adverse health impacts.

## Effects of exposure on vulnerable groups

In a novel study of air pollution and early life metabolomic changes oxylipin profiles in early life were quantified as targets of the pregnancy exposome in association with in utero exposure to particulate matter (PM) air pollution (Martens et al, 2015). Thirty-seven specific oxylipins reflecting specific metabolic pathways were quantified in 197 cord blood plasma samples form the ENVIRONAGE birth cohort. Principal component analysis and multiple regression models were applied to associate oxylipin pathways as well as individual metabolites with in utero PM2.5 exposure, while adjusting for new-borns gender, gestational duration, maternal age, maternal smoking status, maternal BMI, and cord blood total cholesterol and HDL levels. The in utero exposure to particulate matter was associated with alteration in the cord blood levels of metabolites derived from the lipoxygenase pathways indicating an altered inflammatory state of the new-born at birth induced by air pollution during in utero life. Indoor air pollution levels of households of pregnant women have also been quantified. No significant associations between metabolomics changes and indoor levels of NO2, benzene or toluene concentrations were found.

One large study was done to examine possible associations between the traffic pollution situation at the home address spontaneous preterm delivery, children born small for gestational age (SGA) and pregnancy-induced hypertensive disorders (Olsson et al, 2015; 2015b). Data for the Greater Stockholm Area was collected from the Swedish Medical Birth Register to construct a cohort based on all pregnancies conceived between July 1997 and March 2006, in total 100,190 singleton births. The pregnancy average nitrogen oxide, NOx, levels and annual mean daily vehicles at the home address were used as exposure variables. Mixed-model logistic regression was performed to assess any associations between exposure and outcome. There was an association between elevated traffic pollution exposure during pregnancy and pregnancy-induced hypertensive disorders. A 10 µg/m3 increase in the pregnancy average NOx level at the home address resulted in an odds ratio of 1.17 (95% CI 1.10 to 1.26). There was also a tendency of a higher risk of spontaneous preterm delivery in relation to higher levels of NOx. There was no evidence of an association between vehicle flow, the cruder indicator of traffic pollution, and the studied outcomes in this study. A second phase of the study investigated the association with traffic-related particles, and found similar associations. Exposure to elevated levels of PM10 from traffic in early pregnancy was associated with an increased risk of pre-eclampsia and gestational hypertension, and appears also to affect the risk of preterm delivery.

One study was undertaken to study possible associations between air pollution exposure during pregnancy and infancy and asthma medication in childhood in a large cohort (Olsson et al, 2015c). Using register data the researchers followed all singleton children in the Greater Stockholm Area born between July 1st 2000 and Nov 30 2005 up to the age of six. Traffic flow average and modelled annual mean concentration of NOx at home address were used to estimate exposure. The outcome of interest was dispensed asthma medication between five and six years of age. Multiple logistic regression was used to estimate the association between vehicles exhaust exposure and asthma medication. The largest adjusted model includes 66,572 children with valid data. The prevalence of any asthma medication between five and six years of age was 9.7%. We observed no increased risk associated with any of the proxies for vehicle exhaust. Quite opposite, the odds ratios for a 10μg/m3 increase in NOx were 0.92, (95% CI=0.87-0.96), 0.91, (95 % CI=0.86-0.96) and 0.94, (95% CI=0.82-1.09) during gestation, infancy, and during infancy among those who moved during gestation, respectively. However considerable heterogeneity exists also in earlier findings of traffic air pollution exposure and childhood asthma. Preterm birth is usually adjusted for in studies of air pollution and asthma. This may lead to bias in the estimated association because elevated air pollution exposure during pregnancy may also lead to pre-eclampsia and a higher prevalence of preterm births, which in turn may increase the risk of respiratory problems and childhood asthma.

The triggering effect of temperature on the risk of preterm birth during the warm and cold season was investigated in Flanders, Belgium (Cox et al, 2015). We used a quasi-Poisson model on 446,110 singleton natural deliveries between 1998 and 2011. The researchers accounted for the daily pregnancies at risk and their gestational age distribution and we allowed for delayed and non-linear temperature effects. Relative to the median value of 12.8°C, the risk of preterm birth in the warm season increased by 14.1% (95% CI: 4.7-24.2) when minimum temperature up to 3 days before delivery (lag 0–3) exceeded the 95th percentile (17.9°C). The corresponding estimate for maximum temperature was not significant (2.6%, 95% CI: -8.7-15.4). In the cold season, a significant effect of low minimum temperature on the risk of preterm birth two days before birth (lag 2) was observed, but cumulative cold effects were not significant. Even in a temperate climate, ambient temperature might trigger preterm delivery, suggesting that pregnant women should be protected from temperature extremes.

A similar study was performed with the aim to explore the potential association between both heat and cold during late pregnancy and an increased risk of preterm birth in the northern location of Stockholm, Sweden (Vicedo et al, 2015). Data on all deliveries by women residing in the greater Stockholm area from 1998 to 2006 were extracted from the Swedish Medical Birth Register. The analysis was restricted to all singleton births whose labour had not been artificially induced and whose reported gestational age was between 22 and 42 weeks. During the study period, daily air temperature and humidity data were derived from the monitor located in Central Stockholm and managed by the City of Stockholm Environment and Health Administration. Non-linear and delayed effects of mean temperature on the risk of preterm birth were explored. The risk of preterm birth increased by 4%–5% when the mean temperature reached the 75th percentile (moderate heat) four weeks earlier (reference: the annual median value), with a maximum cumulative risk ratio of 2.50 (95% confidence interval: 1.02–6.15). Inconsistent associations were obtained for cold and more extreme heat. Exposure to moderately high temperatures during late pregnancy might be associated with an increase in risk of preterm birth in Stockholm.

## Health impact assessments

One health impact assessment for Grenoble and Lyon took as starting point how risk assessment studies often ignore within-city variations of air pollutants (Morelli et al, 2016). The objective for this HIA was to quantify the risk associated with fine particulate matter (PM2.5) exposure using fine-scale air pollution modelling instead of relying on background monitoring stations.  In Grenoble and Lyon areas (0.4 and 1.2 million inhabitants, respectively) in 2012, PM2.5 exposure was estimated on a 10x10 m grid by coupling a dispersion model to population density. In addition to the most common outcomes in HIA, mortality, also lung cancer and term low birth weight were included as health outcomes. The total numbers of cases attributable to air pollution were estimated, but also numbers after stratifying smaller areas according to the European Deprivation Index. The WHO guideline of 10 μg/m3 was used as reference level. Median PM2.5 levels were 18.1 and 19.6 μg/m3 in Grenoble and Lyon urban areas, respectively, corresponding to 114 (5.1% of total, 95% confidence interval, CI, 3.2–7.0%) and 491 non-accidental deaths (6.0% of total, 95% CI 3.7–8.3%) attributable to long-term exposure to PM2.5, respectively. In Grenoble, 6.8% of incident lung cancer cases were attributable to air pollution (95% CI 3.1–10.1%). Attributable term low birth weight cases represented 23.6% of total cases (9.0–37.1%) in Grenoble and 27.6% of cases (10.7–42.6%) in Lyon. The health impact was lower by 8 to 20% when estimating exposure through background stations. Risk assessment studies relying on background stations to estimate air pollution levels may underestimate the attributable risk. Pregnant woman and their unborn children belong to the vulnerable groups often not included in health impact assessments. The highest attributable fraction was in this case estimated for term low birth weight. Moreover, since the window of sensitivity is likely to correspond to pregnancy, after a decrease in PM2.5 concentration the beneficial effects will be achieved quickly. For outcomes as mortality and lung cancer the beneficial effects will take more time to reach.

In a detailed health impact assessment current and future projected ozone induced mortality in Europe is estimated (Orru et al, 2015). The study aims to quantify effects of both short- and long-term exposure on mortality and assess the calculation's sensitivity to different factors and assumptions. European near surface ozone concentrations were modelled at a grid size of 50x50 km using the chemistry-transport model MATCH4 with anthropogenic precursor emissions from RCP4.5. For the ozone the global climate model EC-EARTH, is used as input of the regional climate model RCA4. The current climate as 1991–2000 was compared to the future climate as 2046–2055. For the health impact assessment the mean of ozone daily 1-hr maximum concentrations from April to September and the sum of means over 35, 25, 10 ppb (daily maximum 8-hour) were calculated. For country impact calculations the population average exposures were retrieved. The data on mortality was attained from WHO's European Health for All Database. The gridded population data for Europe in 2010 and 2050 was taken from the European projects INTARESE and HEIMTSA, where the current and future gridded population data sets have previously been constructed. To see main driver of future mortality change, only one of the drivers were modified and other factors kept constant in each step. In order to describe the ozone effects due to short-term and long-term exposures, the relative risk coefficients from European APHEA2 and American cohort studies are applied. The future ozone levels are expected to increase due to decrease in ozone precursors and increase due to climate change. However, taken together the near ground ozone levels are expected to be smaller in the future. Exposure to near-surface background ozone in Europe is currently expected to cause 44,000 premature deaths in Europe annually, from which 26,000 due to short-term exposure (assuming a cut-off for effects at 25 ppb). The results are most sensitive to implemented cut-off levels, if there is no or a low threshold, at least for long-term exposure, the true health impacts could be bigger than previously assessed.

In a European wide health impact assessment the range of future impacts from high temperatures was studied using 8 different climate models from the RCP45 and RCP85 climate scenarios on a 50x50km grid over Europe (Åström et al, 2015). To estimate the future burden from elevated temperatures two time periods were compared. 1981-2010 represented the present day climate and 2035-2065 represented the future period. To estimate how the temperatures will add to the risk we extracted relative risk ratios (RRs) from a multi-city study within the EU-funded PHASE project including 9 European cities. In this study the increase in mortality was estimated by looking at the change (relative risk) when temperatures increased from the 75th to the 99th percentile of summer temperatures (April to September). For the HIA the increase in mortality would is assumed to be linear between the 75th and 99th percentile and beyond. The increase in mortality is estimated using the mean, highest and lowest risk functions from the 9 cities in the study. This gives a range of impacts which show how much different levels of adaptation might impact the future health burden. The population data used in the study was a 50x50km gridded dataset compiled by the INTARESE and HEIMSTA project. With the population data it is possible to calculate the expected daily mortality by using standardized mortality rates for the different countries from Eurostat database. The population data is limited to 29 countries in the European region. The estimate impacts from elevated temperatures are calculated for each grid cell individually and then weight together based on population density to get a representative impact for each country.

The general pattern of the estimations is that around 1% of total mortality in the different countries is results of elevated temperatures when using the mean of the RRs. For the future period the range of impacts from the different models shows large variation. The results for RCP45 shows an estimated annual mortality related to heat in the different countries ranging from 1.3 to 4.3% when using the mean of the relative risks. In RCP85 the estimates range from 1.3 to 5.5% of the annual mortality. Thus, the lower limit from both RCP45 and RCP85 is very similar for many countries whereas the higher limit differs more.

One of the health impact assessments focussing on climate change aims to investigate how changing demographics and the size of the vulnerable population, as well as the role of adaptation, may influence the future temperature-related mortality (Oudin et al, 2015). The future summertime temperatures were extracted for 16 sites in the Stockholm County from the RCA4 climate realization based on EC-EARTH and RCP4.5. The temperature-mortality relationship during the summer months of the baseline period 2000-2009 was established thorough a time series approach, using a generalised additive model and stratified according to age and pre-existing chronic disease. Temperatures below the 75th percentiles in the baseline and future temperature distributions were assigned the value 0 and temperatures corresponding to the 99th percentiles were assigned as 1. Temperatures between the 75th and the 99th percentile as well as temperatures above were assigned values assuming a linear increase in mortality between the 75th and 99th percentile and beyond. The number of times temperatures increased from the 75th to the 99th percentile, increased, on average, with 68% between the periods. By 2050 the average annual excess deaths due to high summer temperatures are in Stockholm County, according to our scenarios, expected to increase to 269, as compared to 117 in 2010. The role of changes in demographics, vulnerability and adaptation and the impact such changes would have on yearly excess deaths as compared to the main scenario. Adaptation would reduce heat related mortality by 20% if mortality rates among the elderly are to converge to mortality rates among the younger ages. Decreasing relative risks (RR) associated with exposure to high summer temperatures would reduce summer mortality by 22%. Assuming a continued linear reduction in mortality rates would reduce summer mortality by 11%. This assessment shows that in addition to accounting for different climatic scenarios it is important to take into account the different effects that adaptation, changing demographics and changing vulnerabilities will have when projecting future impacts of temperature on mortality.

One health impact assessment for Greater Stockholm aims at investigating the impact on future population exposure of climate change versus of emission changes related to potential changes in road traffic solutions (Andersson et al, 2015). For nitrogen oxides (NOx) and exhaust particles (PM-exhaust) the impacts of three mitigation strategies are compared - a motorway bypass outside Stockholm, no exhaust emissions (electric fleet), and a fossil free vehicle fleet - with climate change effects on the exposure. For ozone changes in exposure due to European and local emissions are compared. Both effects of large-scale global climate change and fine-scale local emission change are taken into account. The effects on NOx and PM exposure are investigated using a Gaussian dispersion model at 100m resolution. For surface ozone, nested model calculations from global (50 km) to local scale (1 km over Stockholm) are used. An Eulerian chemistry-transport model (MATCH) was used to analyse impacts on the air quality 2050 of climate change compared with changing European and local emissions. MATCH includes ozone- and aerosol-forming photochemistry with 60 species. In the FFF fleet it is assumed that all diesel and gasoline vehicles would be electric and the remaining emissions (difference between no exhaust and fossil free) are due to existing ethanol, biogas and biodiesel vehicle emissions. The planned bypass will be much less beneficial for reduced population exposure compared to the FFF scenario and that climate change alone has negligible effect on the exposure. Future ozone exposure in Stockholm will decrease due to projected lower European precursor emissions. Exchanging all vehicles to electric will cause increasing ozone exposure in Stockholm due to less NOx emissions, but this effect is very small compared to the projected decrease of ozone exposure in Stockholm due to projected lower European precursor emissions.

## Policy contribution

The major part of our scientific outcomes presented above are relevant with regards to policy in the field of environment and health. Here some of the important findings are highlighted:

In one study estimate that around 1% of total mortality in the studied European countries is a result of high temperatures. For the future climate as the period 2046–2055, the results for RCP45 shows an estimated annual mortality related to heat in the different countries ranging from 1.3 to 4.3% when using a mean of the city specific relative risks from a recent European study. This range may indicate how much difference the level of adaptation can make.

Until now, most assessments of health impacts of heat under a warmer climate have focussed on excess mortality. The studies of associations between extreme temperatures and birth outcome indicate that also such outcomes as preterm birth are important to consider. At least the results for extreme heat in Flanders are in line with findings from southern European cities like Rome and Valencia.

Projections of urban scale climate change show substantial warming in the cities of Brussels and Paris, 1.6°C and 1.8°C respectively, until the 2050s horizon. A decrease in daytime urban heat island (UHI) is projected in summer connected to drying soils over rural areas. However a wintertime increase in the nocturnal UHI intensity is projected due to decreased wind speeds, which may result in an increase in particulate matter locally in urban areas. Thus, it will be very important for policy makers to make sure that the reduction in anthropogenic emissions will continue to counteract this climate penalty.

Using a four-model ensemble it is predicted that European anthropogenic emission decrease until 2050 which will cause a strong improvement in air quality, in general having a much stronger impact than climate change on the air quality in Europe. However, this conclusion is dependent on the resolution of the emission inventory and models applied. When describing Paris at higher resolution, climate change turns out to be the most influential factor for changes in maximum ozone.

In one health impact assessment current and future projected ozone induced mortality in Europe is estimated. European near surface ozone concentrations were modelled at a grid size of 50x50 km using the chemistry-transport model MATCH4 with anthropogenic precursor emissions from RCP4.5. The current climate as 1991–2000 was compared to the future climate as 2046–2055. The future ozone levels are expected to increase due to decrease in ozone precursors and increase due to climate change. However, taken together the near ground ozone levels are expected to be smaller in the future. The results are very sensitive to implemented cut-off levels, if there is no or a low threshold, at least for long-term exposure, the true health impacts of ozone could be bigger than previously assessed.

One health impact assessment for Grenoble and Lyon compared results obtained using exposure data from urban background monitoring stations with results built on dispersion modelling allowing for within-city variations of concentrations. The health impact was lower by 8 to 20% when estimating exposure through background stations instead of dispersion modelling, questioning the approach classically used in France for the assessment of risk associated with fine particulate matter. The highest proportion of cases attributed to air pollution exposure was in this case estimated for term low birth weight, an outcome usually not included in similar health impact assessments. A large epidemiological study in Stockholm adds support for local traffic-related pollutants to be important for an increased risk of adverse birth outcomes. The health impact assessment study indicated that, in Grenoble and Lyon, areas of moderate to high social deprivation were those suffering most of exposure to fine particulate matter.

For Stockholm the health impacts of several different mitigation scenarios have been estimated. Moving traffic on a major high-way into a tunnel in Stockholm would decrease the health impacts due to reduced traffic pollution exposure in the residents, but the positive effect is cancelled due to the increased exposure among tunnel users. Exchanging fossil fuelled vehicles by electric would reduce the particulate matter exposure, but it would increase the ozone exposure. However, the positive benefit of climate change on surface ozone in Stockholm is projected to be stronger.

The impact of enforced low emission zones (LEZ) depends on the strictness of the regulation as well as the size of the regulated area, and that it is most effective for health relevant components (such as soot) contained in PM10. It means that even though the impact on PM10 may be small, the impact on health can be stronger since the LEZ reduce exhaust particles that may have the most adverse health impacts. In order to make a proper evaluation of control policies, it is very important to monitor the impacts on traffic intensities and vehicle composition both inside and outside the zone. In addition, the evaluation of LEZ by monitoring of EC and/or OC in PM10 is recommended.

The ISSeP study of indoor levels of VOC and NO2 in 200 houses of the Belgian ENVIRONAGE birth cohort found wide distributions of concentrations between the houses indicating that exposure to indoor pollution is unequal and that individual exposure data are particularly relevant when dealing with indoor air and health impact.

Most of the parents have integrated the necessity to air the house by opening windows every day. On the other hand the period surrounding the birth is an intensive renovation period with subsequently VOC emissions.

The results of this measurement campaign in 212 houses supply references to the authorities on the air quality inside the housing stock in Belgium. Determinants that contribute to enhance concentrations of carcinogens inside houses of new-borns were highlighted.

Results from this study provide references relevant for policy making and to identify recommendations to prevent exposure to harmful pollutants during the period surrounding birth.

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## Table of deliverables

| No. | Title | Ownership & Status |
|---|---|---|
| 1 | Minutes of the WP1, WP2 and WP3 workshops | WP1/2/3<br>on time |
| 2 | Mid-term report on policy relevance and knowledge transfer | Coordinator<br>on time |
| 3 | Report on local air quality effects of low emission zones in European cities | WP1<br>on time |
| 4 | Report on the effect of future climate and control policies on air quality in European cities | WP1<br>on time |
| 5 | Report on exposure models adjusting for building, weather and other factors modifying actual exposure | Draft is circulating |
| 6 | Manuscript on air pollution, temperature extremes and preterm birth in Sweden and Belgium | WP3<br>on time, but three separate papers |
| 7 | Manuscript on air pollution exposure and development of childhood asthma | WP3<br>on time |
| 8 | Manuscript on molecular epidemiology of mitochondrial function in relation to indoor and outdoor air pollution levels | WP3 Readjusted to Martens et al. Altered neonatal cord blood lipidome ... (see publication list) |
| 9 | Report on guidelines for health impact assessment of future climate and air pollution scenarios | Draft is circulating |
| 10 | Project summary report for end-users | WP1-4 December 2015 |

# 5. DISSEMINATION OF RESULTS AND KNOWLEDGE TRANSFER

## Participation in scientific events; posters and presentations

Andersson Camilla, Magnuz Engardt and Camilla Geels. Changes to the European particle composition during the 21st century. Oral presentation at the Air Quality Conference in Garmisch-Partenkirchen, March 25-28, 2014.

Cyrys J. “LEZ in München & Augsburg - Analyses of impacts based on measurements of PM10”. Oral presentation at ACCEPTED workshop “Low Emission Zones – how to estimate the air quality benefits”. Stockholm, 17-18 June, 2014.

Cyrys J: “LEZ in Berlin - Analyses of impacts based on measurements of BS and PM2.5. Oral presentation at ACCEPTED workshop “Low Emission Zones – how to estimate the air quality benefits”. Stockholm, 17-18 June, 2014.

Cyrys J, V. Fensterer, H. Küchenhoff, B. Bauer, H.-E. Wichmann, S. Breitner, A. Schneider, A. Peters: The impact of low emission zone and heavy traffic ban in Munich on the reduction of PM10 in ambient air. Oral presentation at the 9th International Conference on Air Quality, Science and Application, Garmisch-Partenkirchen, 24-28 March 2014

Gu J, Camilla Andersson, Cecilia Bennet, Lars Gidhagen, Kathrin Wolf, Alexandra Schneider, Jens Soentgen, Annette Peters, Josef Cyrys. Modelling of Exposure to Ambient Particulate Matter and NOx in Augsburg, Germany. Poster presentation at European Aerosol Conference 2015, Milan, September 6 -11, 2015.

Cyrys J, J. Gu, V. Deffner, H. Küchenhoff, J. Soentgen, A. Peters: Effects of Low Emission Zones (LEZ) on air quality in Germany. Oral presentation at European Aerosol Conference 2015, Milan, September 6 -11, 2015.

Cyrys J, J. Gu, V. Deffner, H. Küchenhoff, J. Soentgen, A. Peters: Analyse der Wirksamkeit von Umweltzonen in drei deutschen Städten: Berlin, München und Augsburg. Oral presentation at VDI seminar ”Neue Entwicklungen bei der Messung und Beurteilung der Luftqualität”. Nürnberg, 20.-21. October 2015.

Cyrys J, J. Gu, V. Deffner, H. Küchenhoff, V. Maier, J. Soentgen, A. Peters: Effects of Low Emission Zone in three German cities. Poster presentation at the ACCEPTED final meeting. 23 October 2015, Brussels.

Christer Johansson, Josef Cyrys, Jens Soentgen, Jianwei Gu, Camilla Andersson, Cecilia Bennet & Lars Gidhagen: Air pollution exposure in European cities: Effects of traffic policy. Oral presentation at the ACCEPTED final meeting. 23 October 2015, Brussels.

Cox B., Vicedo-Cabrera A.M., Gasparrini A., Roels H.A., Martens E., Vangronsveld J., Forsberg B., Nawrot T.S. (2015) Ambient temperature as trigger of preterm delivery in a temperate climate. . ACCEPTED Final Meeting. Brussels, Belgium, 23 October, 2015. Poster Presentation

Cox B., Vicedo-Cabrera A.M., Gasparrini A., Roels H.A., Martens E., Vangronsveld J., Forsberg B., Nawrot T.S.  Ambient temperature as trigger of preterm delivery in a temperate climate. European Congress of Epidemiology – Healthy Living. Maastricht, Netherlands, 25-27 June, 2015. Poster Presentation.

Cox B., Vicedo-Cabrera A.M., Gasparrini A., Roels H.A., Martens E., Vangronsveld J., Forsberg B., Nawrot T.S. (2015) Ambient temperature as trigger of preterm delivery in a temperate climate. 1st Young Researchers Conference on Environmental Epidemiology- ISEE Europe. Barcelona, Spain, 20-21 October, 2014. Oral Presentation.

Delcloo Andy, Camilla Andersson, Bertil Forsberg, Tim Nawrot, Myrto Valari for the ACCEPTED Project, ACCEPTED; an Assessment of Changing Conditions, Environmental

Policies, Time-activities, Exposure and Disease. Poster at the 33rd International Technical Meeting on Air Pollution Modelling and its Application, 26-30 August, 2013, Miami, Florida USA.

Delcloo A, R. De Troch, O. Giot, R. Hamdi, A. Deckmyn and P. Termonia, ITM, Montpellier, France, 4 - 8 May 2015.Future climate and air quality of the Brussels Capital Region for the 2050's under A1B scenario, ITM, Montpellier, France, 4 - 8 May 2015.

Hamdi R. Assessment of three dynamical urban climate downscaling methods: Application for Brussels and Paris. 3rd International Lund Regional-Scale Climate Modelling Workshop 21st Century Challenges in Regional Climate Modelling Lund, Sweden, 16-19 June 2014.

Hamdi R, Hans Van de Vyver, Rozemien De Troch, Piet Termonia, Andy Delcloo: Assessment of three dynamical urban climate downscaling methods, 33rd International Technical Meeting on Air Pollution Modelling and its Application, 26-30 August, 2013, Miami, Florida USA.H

Hamdi R. Assessment of three dynamical urban climate downscaling methods: Brussels's future urban heat island under an A1B emission scenario, The International Conference on Regional Climate - CORDEX 2013, 4-7 Nov, Brussels, Belgium.

Markakis, K., Valari, M., Powaga, E., Collignan, B., Perrussel. O., Joly, F.: High-resolution modelling of human exposure for the mid-21st century in Paris, France under changed climate, emissions and building stock evolution. Oral presentation at the "international conference of atmospheric sciences and applications to air-quality", 11-13 November, 2015, Kobe, Japan.

Markakis, K., Valari, M., Engardt, M., Lacressoniere, G., Vautard, R., Andersson, C.: Mid-21st century air-quality at the urban scale: case studies for Paris and Stockholm. Poster presentation at the "Air quality conference", 14-16 March, Milan, Italy.

Martens D.S., Gouveia-Figueira S., Madhloum N., Janssen B. G., Plusquin M., Forsberg B., Nording M. L., Nawrot T.S. (2015). Altered neonatal cord blood lipidome in association with exposure to particulate matter in the early life environment. ACCEPTED Final Meeting. Brussels, Belgium, 23 October, 2015. Oral Presentation.

Martens D.S., Gouveia-Figueira S., Madhloum N., Janssen B. G., Plusquin M., Forsberg B., Nording M. L., Nawrot T.S. (2015). Altered neonatal cord blood lipidome in association with exposure to particulate matter in the early life environment. 2nd Early Career Researchers Conference on Environmental Epidemiology. Utrecht, Netherlands, 2-3 November, 2015. Oral Presentation.

Martens D.S., Gouveia-Figueira S., Madhloum N., Janssen B. G., Plusquin M., Forsberg B., Nording M. L., Nawrot T.S. (2015). Altered neonatal cord blood oxylipidome in association with exposure to particulate matter in the early life environment. Methods in Epidemiology Symposium. Leuven, Belgium, 17 September, 2015. Oral Presentation.

Martens D.S., Gouveia-Figueira S., Madhloum N., Janssen B. G., Plusquin M., Forsberg B., Nording M. L., Nawrot T.S. (2015). Altered neonatal cord blood lipidome in association with exposure to particulate matter in the early life environment. European Congress of Epidemiology – Healthy Living. Maastricht, Netherlands, 25-27 June, 2015. Oral Presentation.

Martens D.S., Gouveia-Figueira S., Madhloum N., Janssen B. G., Plusquin M., Forsberg B., Nording M. L., Nawrot T.S. (2015). Altered neonatal cord blood lipidome in association with exposure to particulate matter in the early life environment. PIC2015 – the 14th International Congress on Combustion By-Products and Their Health Effects. Umeå, Sweden, 14-17 June, 2015. Oral Presentation.

Martens D., Gouveia-Figueira S., Madhloum N., Janssen B. G., Plusquin M., Forsberg B., Nording M. L., Nawrot T.S. (2014). Cord blood lipidomics in association with maternal current and past tobacco exposure. 1st Young Researchers Conference on Environmental Epidemiology- ISEE Europe. Barcelona, Spain, 20-21 October, 2014. Poster Presentation.

Moldanova Jana, Lin Tang, Malin Gustafsson, Håkan Blomgren, Tomas Wisell, Erik Fridell, Bertil Forsberg. Emissions from traffic with alternative fuels - impacts on exposure and health in 2020. Accepted Final meeting, Brussels, October 2015.

Morelli X, Camille Rieux, Josef Cyrys, Bertil Forsberg, Rémy Slama: Air pollution, health and social deprivation in an urban setting: a fine-scale risk assessment. Oral presentation at the ACCEPTED final meeting. 23 October 2015, Brussels.

Olsson D and Forsberg B. Associations between particulate matter and adverse pregnancy outcomes. Accepted Final meeting, Brussels, October 2015.

Olstrup Henrik, Christer Johansson, Bertil Forsberg. Estimating mortality attributed to ambient particle concentrations – use PM mass or soot? Accepted Final meeting, Brussels, October 2015. Poster presentation.

Orru Hans, Boel Lövenheim, Christer Johansson, Bertil Forsberg. Potential health impacts of Stockholm bypass due to changes in air pollution exposure. Accepted Final meeting, Brussels, October 2015.

Orru H, Åström C, Andersson C, Ebi K and Forsberg B. Impact of climate and emission change on ozone induced mortality in Europe with combining effect of heat. ISEE conference 2015.

Orru H, B. Lövenheim, C. Johansson, B. Forsberg: Assessing the health benefits of decreased population exposures versus disbenefits of increased driver exposures in an 18 km long high-way road tunnel by-pass in Stockholm. Proceedings of Abstracts 9th International Conference on Air Quality, Science and Application, Garmisch-Partenkirchen, 24-28 March 2014

Orru H, C. Åström, C. Andersson, K. L. Ebi, B. Forsberg: Impact of climate and emission change on ozone induced mortality in Europe with combining effect of heat. In "Human Health in the Face of Climate Change: Science, Medicine, and Adaptation" Conference. Barcelona, 14-15 May, 2015.

Orru H, B. Lövenheim, C. Johansson, B. Forsberg: Estimated health impacts of changes in air pollution exposure associated with the planned by-pass Förbifart Stockholm. NOSA Aerosol Symposium. Stockholm, 16-17 October 2013.

Orru H, B. Lövenheim, C. Johansson, B. Forsberg: Health impacts of traffic exhaust in road tunnels – a case of planned 18 km long high-way road tunnel by-pass in Stockholm. PIC2015 – the 14th International Congress on Combustion By-Products and Their Health Effects. Umea, 14-17 June 2015.

Powaga E, B. Collignan, Modelling of outdoor pollutant transfers in buildings. ACCEPTED final meeting, Brussels, October 2015. Poster presentation.

Vicedo-Cabrera Ana, David Olsson, Bertil Forsberg. Exposure to seasonal temperatures during the last month of gestation and the risk of preterm birth in Stockholm, Accepted Final meeting, Brussels, October 2015. Poster presentation.

WP2 workshop in Paris in November 2013: The design of the monitoring campaign, the preliminary sampling and analysis tests were presented during the WP2 workshop in Paris.

## Interactions and joint activities

Some of the common activities within the ACCEPTED Project have primarily been initiated to reach out to various stakeholders in order to enable communication and interaction. Among these activities we can mention the ACCEPTED Website, the ACCEPTED Newsletter, the seminar on Low Emission Zones LEZ (arranged in Stockholm in June 2014) and the Final Project meeting (held in Brussels in October 2015).

SMHI has had interactions with another large project, IMPACT2C funded by EU (FP7). The collaboration has led to mutual scientific publications.

The City of Stockholm and Umeå University have collaborated with The Swedish Transport Administration on a health impact assessment of a planned by-pass road in Stockholm.

Umeå University has participated in the EU-funded PHASE Project on extreme weather events and been able to use results from PHASE in the health impact assessments on future heatwaves.

The exposure model developed during the ACCEPTED project is a key component in the MouVIE project of CNRS, which is a 5 years French project co-funded by the PSA Peugeot Citroen and Renault and the University Pierre et Marie Curie to study mobility and life quality in urban areas (http://www.upmc.fr/fr/salle_de_presse/chaire_upmc_psa_renault.html). As part of the project a three-year PhD thesis will focus on methods of the exposure model validation based on personal exposure data from cohort studies using portable monitors.

Thanks to ACCEPTED, ISSeP is now able to provide experience and references in indoor air pollutant analysis. Since ISSeP is a Public Scientific Institute, we are approached by the regional government to participate in a consortium dealing with a better understanding of indoor air pollution in homes. This consortium will prepare a roundtable (scheduled for November 2016) on indoor air quality prevention and guidelines setting.

## Information / technology transfer

Information has been transferred to many different groups, see above about presentations and interactions.

At the Seminar on low emission zones, held in Stockholm 2014, there were demonstration of control policies and evaluations, with e.g. a presentation of the Augsburg modelling results.

Also at the Final meeting in Brussels there were reports, with e.g. a presentation of the future scenarios of climate and control policies, and about low emission zones and other enforced control policies.

An article about experiences from ACCEPTED is going to be included in the SMHI annual report 2015 (a popular science book).

ISSeP acquire experiences in questionnaire design, indoor passive sampling and analyses, indoor air quality data analysis.

## Outreach to the general public

The ACCEPTED Newsletter has been sent out frequently during the project and a last issue is planned to summarise the final report; thus there will be in total seven issues published (July 2013, September 2013, March 2014, August 2014, February 2015, June 2015 and finally, in January 2016.

A popular science article has been published on the SMHI website and an article will be included in the SMHI annual report 2015 (popular science book).

Results from the Stockholm studies of associations between air pollution exposure and birth outcome have been reported in Swedish news media.

Results from the health impact assessment of a by-pass road in Stockholm have been reported in Swedish news media.

Interviews were given a journalist at the Daily Science at the final meeting in Brussels:

Biomarqueurs de l'inflammation dans le sang de cordon (Martens D., Nawrot T.S.). In: Les bébés n'aiment pas le réchauffement climatique, Christian Du Brulle, Daily Science. http://dailyscience.be/2015/10/28/les-bebes-naiment-pas-le-rechauffement-climatique (Accessed 16 November 2015)

Les nuits torrides d'été augmentent le risque de naissance prématurée (Cox B., Nawrot T.S.). In: Les bébés n'aiment pas le réchauffement climatique, Christian Du Brulle, Daily Science. http://dailyscience.be/2015/10/28/les-bebes-naiment-pas-le-rechauffement-climatique (Accessed 16 November 2015)

Tim Nawrot also participated in television interviews (Thursday December 10th, program 'Koppen' and a radio interview ('nieuwe feiten Radio 1, Wednesday 9th of December '15), and ascientific interview on The "Omics" Approach to Science Has Emphasized the Nature Side of the Nature/Nurture Divide see: http://www.genengnews.com/gen-articles/the-naturome-awaits-the-nurturome/5639/

Results of the Belgian indoor air pollutants levels during early childhood will be presented at the roundtable on indoor air quality in November 2016 organized by the Walloon Government.

# 6. USES AND IMPACTS

## Impact statement

See policy contribution on page 16-17.

## New technologies

The exposure model developed during the ACCEPTED project by CNRS is a major research activity of the Laboratoire de Météorologie Dynamique devoted to climate and air-pollution impact on humans, the society and the environment. Further funding to continue the development of the on-line coupling of outdoor and indoor air-quality as well as time-activity data is considered.

CSTB has adapted the ventilation model SIREN to take into account transfer between indoor and outdoor for Ozone and PM2.5.

## Follow up activities and plans for further exploitation of the results

The ACCEPTED Website will continue to be updated with new publications and presentations at least during 2016.

Scientific publication will continue after the project has ended in December 2015. Presentations at scientific conferences will also continue take place after the project has ended.

The Swedish project partners, Umeå University, SMHI and City of Stockholm, plan for a seminar at the Swedish EPA to take place after the end of the project with the aim to summarise the results of the project relevant to the Swedish authorities and policy makers.

SMHI plan to further disseminate the results at future stake holder meetings.

A scientific paper with a detailed technical and scientific description of the exposure models' functionality is under preparation. The goal is that the source code and all the necessary input data will be uploaded on the CHIMERE model site (http://www.lmd.polytechnique.fr/chimere/) and will be distributed under open source license. Several scientific papers will follow discussing different applications of the models.

# 7. DATA MANAGEMENT AND TIMELINE FOR OPEN ACCESS

The goal has been that methods and results are published in peer review papers och reports that can easily be accessed. The Project web is updated at least during 2016.

Each project partner is responsible to judge if data in a meaningful form can be linked to relevant databases, or already is part of such data bases (e.g. meteorological data). National funding agencies may have specific requirements.

Swedish and Belgian epidemiological data (used in WP3) must be kept by the research team according to the ethical approval, but descriptive statistics can be requested from the responsible partners.

# 8. EXPLANATION OF THE USE OF RESOURCES

Due to more workshops and meetings than originally planned, the costs for Travel and meetings became higher than in the initial budget, and salary costs had to be kept lower.

| **Financial report** | **Partner:** | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | **TOTAL** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Exchange rate:** | 9,242 | 9,242 | 9,242 | | | | | | | | | |
| | **Date of rate:** | 151201 | 151201 | 151201 | | | | | | | | | |
| | **Currency of origin:** | SEK | SEK | SEK | | | | | | | | | |
| Salaries (costs, person and function) | | 78 740 | 65 858 | 35 202 | 152 167 | 50 000 | 94 250 | | 77 618 | 62 869 | | 42 363 | 659 067 |
| Sub-contractors (costs, sub-contractor and function) | | 0 | 0 | 0 | 0 | 0 | 0 | | 14 174 | 0 | | | 14 174 |
| Travel, meeting and conference costs | | 37 448 | 6 652 | 1 298 | 12 728 | 5 000 | 4 000 | | 2 829 | 4 136 | | 2 655 | 76 746 |
| Data costs | | 576 | 0 | 0 | 0 | 0 | 0 | | | 0 | | | 576 |
| Printing and publication costs | | 776 | 0 | 0 | 0 | 0 | 0 | | | 0 | | | 776 |
| Expenses (cost/type) | | 0 | 0 | 0 | 35 075 | 15 500 | 0 | | 5 855 | 0 | | 10 361 | 66 791 |
| Other Overhead | | 30 549 | 25 378 | 12 775 | 9 998 | 2 500 | 0 | | 4 019 | 66 013 | | 2 274 | 153 506 |
| **TOTAL COST** | | **148 089** | **97 888** | **49 275** | **209 968** | **73 000** | **98 250** | | **104 495** | **133 018** | | **57 653** | **971 636** |

1. Umeå University (coordinator), Bertil Forsberg
2. Swedish Meteorological and Hydrological Institute (SMHI), Camilla Andersson
3. City of Stockholm, Christer Johansson
4. Hasselt University, Tim Nawrot
5. Royal Meteorological Institute of Belgium (RMI), Rafiq Hamdy
6. University of Augsburg, Josef Cyrus
7. Centre National de la Recherche Scientifique (CNRS), Myrto Valari
8. Institut national de la santé et de la recherche médicale (INSERM), Rémy Slama
9. Centre Scientifique et Technique du Bâtiment (CSTB), Bernard Collignan, Emilie Powaga
10. AIRPARIF, Cecile Honoré
11. Institut scientifique de service public (ISSeP), Suzanne Remy

# 9. ANNEX

## Translations of publishable summary

1. Dutch
2. French
3. German
4. Swedish

## Deliverables

1. WP1 meeting minutes, 2013-08-13
   WP2 meeting minutes, 2013-11-22
   WP3 meeting minutes, 2014-03-04
2. Mid-term report, 2014-06-03
3. LEZ report, 2014-06-11
4. Report on the effect of future climate and control policies on air quality in European cities. C Andersson, C Johansson, K Marakakis, M Valari, A Delcloo, R Hamdi
5. Report on exposure models adjusting for building, weather and other factors modifying actual exposure.
6. A) Article; Exposure to seasonal temperatures during the last month of gestation and the risk of preterm birth in Stockholm. A Vicedo, D Olsson, B Forsberg
   B) Draft of article; Ambient temperature as trigger of preterm delivery in a temperate climate. B Cox, A Vicedo, A Gasparrini, E Martens, J Vangronsveld, B Forsberg, T Nawrot
   C) Article; Traffic pollution at the home address and pregnancy outcomes in Stockholm, Sweden. D Olsson, I Mogren, K Eneroth, B Forsberg
7. Manuscript of article; Traffic pollution exposure at home during pregnancy and infancy and childhood asthma medication. D Olsson, L Bråbäck, B Forsberg
8. Draft of article; Altered neonatal cord blood oxylipidome in association with exposure to particulate matter in the early life environment. D Martens, S Gouveia-Figueira, N Madhloum, B Janssen, M Plusquin, C Vanpoucke, W Lefebvre, B Forsberg, M Nording, T Nawrot
9. Report on guidelines for health impact assessment of future climate and air pollution scenarios.
10. Project summary report for end users.

## Newsletters

1st edition, July 2013
2nd edition, September 2013
3rd edition, March 2014
4th edition, August 2014
5th edition, February 2015
6th edition, June 2015

## Posters, final meeting in Brussels, 22nd of October 2015

1. Exposure to seasonal temperatures during the last month of gestation and the risk of preterm birth in Stockholm.
   Ana M. Vicedo-Cabrera, David Olsson, Bertil Forsberg

2. Emissions from traffic with alternative fuels - impacts on exposure and health in 2020.
   Jana Moldanova, Lin Tang, Malin Gustafsson, Håkan Blomgren, Tomas Wisell, Erik Fridell, Bertil Forsberg

3. Ambient temperature as trigger of preterm delivery in a temperate climate.
   Bianca Cox, Ana M Vicedo-Cabrera, Antonio Gasparrini, Evelyne Martens, Jaco Vangronsveld, Bertil Forsberg, Tim S Nawrot

4. Health Risk Assessment of reducted air pollution exposure when changing commuting by car to bike.
   B. Lövenheim, C. Johansson, L. Wahlgren, H. Rosdahl, J. Salier Eriksson, P. Schantz, P. Almström, S. Berglund, A. Markstedt, M. Strömberg, J. Nilsson Sommar and B. Forsberg

5. Effect of road dust abatement measures on PM10: Vehicle speed and studded tyre reduction.
   M. Norman, I. Sundvor, B.R. Denby, C. Johansson, M. Gustafsson, G. Blomqvist and S. Janhäll

6. Change in future air pollution exposure due to changing emissions & climate Stockholm.
   Christer Johansson, Camilla Andersson

7. Associations between particulate matter and adverse pregnancy outcomes.
   David Olsson, Bertil Forsberg

8. Modelling of outdoor pollutant transfers in buildings.
   E. Powaga, B. Collignan

9. Potential health impacts of Stockholm bypass due to changes in air pollution exposure.
   Hans Orru, Boel Lövenheim, Christer Johansson, Bertil Forsberg

10. Estimating mortality attributed to ambient particle concentrations – use PM mass or soot?
    Henrik Olstrup, Christer Johansson, Bertil Forsberg

11. Effects of Low Emission Zone in three German cities.
    J. Cyrys, J. Gu, V. Deff ner, H. Küchenhoff , V. Maier, J. Soentgen, A. Peters

12. A review of quantitative health impact assessments of ozone and particulate matter under a changing climate.
    V. N. Likhvar, K. Markakis, M. Valari, A. Colette, D. Hauglustaine, S. Medina, M. Pascal, P. Kinney

13. Future health impacts related to ozone, secondary inorganic aerosols and primary PM in Europe – response to projected changes in climate, emissions, demography and building stock.
    C. Geels, C. Andersson, O. Hänninen, P. Schwarze and J. Brandt

# ACCEPTED

Assessment of changing conditions, environmental policies, time-activities, exposure and disease

# Report on Low Emission and Congestion Charge Zones in Europe with impact assessment studies for Augsburg, Munich, Berlin and Stockholm

**Christer Johansson**

Environment and Health Administration, City of Stockholm (also at Department of Applied environmental Science, Stockholm university), Sweden

**Camilla Andersson, Cecilia Bennet & Lars Gidhagen**

Swedish Meteorological and Hydrological Institute, Norrköping, Sweden

**Josef Cyrys, Jens Soentgen & Jianwei Gu**

University Augsburg, Augsburg, Germany

# CONTENTS

# 1 Preface

This report presents result from the ERA-ENVHEALTH project ACCEPTED (Assessment of changing conditions, environmental policies, time-activities, exposure and disease), a transnational research project on environment and health linking scientific advancement to policy and practice.

ACCEPTED aims at improving our understanding of future exposure situations and their impact on health, from an interdisciplinary approach. This project also address the issue of the mitigation strategies that can be used to reduce urbanization and climate change effects on the local urban meteorology and air quality. The ACCEPTED research program is a three year long project that involves 11 different partners from four European countries. The project is coordinated by Umeå University, Sweden (Prof Bertil Forsberg). For further information see http://www.acceptedera.eu/.

Stockholm June 2014

Christer Johansson
Environment and Health Administration
SLB analys
Stockholm, Sweden

# 2 Summary

Low Emission Zones (LEZs) are areas where only vehicles that fulfil certain emission requirements are allowed to enter (without paying). Congestion Charge Zones (CCZs) are areas to which drivers have to pay to enter, irrespective of emission standard. Both LEZs and Congestion Charge Systems (CCSs) may reduce traffic emissions and improve air quality even if they have different purposes (abate traffic emission or congestion). Sometimes LEZ and CCS and other traffic reduction measures have potentially large health benefits. More and more scientific studies have been using monitoring data to assess the potential air quality benefits.

## 2.1 Objectives

The objectives of this study were

- to assess the effect on air quality of a Congestion charging system (CCS) in Stockholm and LEZs (Low Emission Zones) in three German cities (Augsburg, Munich and Berlin).
- to summarize and compare assessments of the effectiveness of LEZs/CCSs in European cities, based on both measurements and modelling.
- discuss the importance of the methodology for demonstrating and detect the effects on air quality of a LEZ/CCS
- discuss the difficulties encountered with attributing changes in air quality to the LEZ, advantages and disadvantages of using different methods, and pose some recommendations for future evaluations.

## 2.2 CCS in Stockholm

### 2.2.1 Statistical analysis

A semiparametric model with first-order autoregressive errors (originally developed for Munich by Fensterer et al. (2014)) was applied to estimate the effect on $NO_x$ and $PM_{10}$ mass concentrations of the CCS in Stockholm. $NO_x$ and $PM_{10}$ at a reference site, as well as other variables including public holidays, wind direction, day of the week and time of the day were included in the statistical model. $NO_x$ and $PM_{10}$ concentrations were modelled separately for winter and summer season. It is implicitly assumed that the differences in concentrations between periods with congestion charge and those without charging at the reference site are due to factors that are not related to the charging (e g changes in vehicle composition leading to lower emissions). But since the charging lead to some increase (few percent) in traffic intensity on the highway at the reference site as it was exempted from charge, the effects of the charge on the concentrations inside the charge cordon may be overestimated.

For $NO_x$ all measurement stations, except the roof-top site at Hornsgatan shows lower concentrations during congestion charge periods as compared to non-charge periods. The reduction ranges from -0.1% to -9.2%. Looking at summer only, all stations (also roof-top stations) show reduced concentrations of $NO_x$ (-6.1% to -18%). For winter, reductions are smaller and one of the street stations and one roof-top station even show increased $NO_x$ concentrations. For $PM_{10}$ the result is less consistent. Two (of three) street sites show reduced concentrations by -8.4% and -10.3%, but one show a small increase (+0.6%). The roof-top station show rather large increase in concentrations by 15.5%.

### 2.2.2 Dispersion modelling

The annual mean $PM_{10}$ and $NO_x$ concentrations and exposures due to local road traffic emissions with and without the CCS were calculated using a diagnostic wind model and a Gaussian dispersion model. The same meteorological data was used, in order to assess the potential effect of the CCS. The CCS lead to 15% reduction in total vehicle transports (km's driven buy all cars), which resulted in reduced vehicle emissions of both $NO_x$ and $PM_{10}$. Reduced congestion lead to a small further reduction in $NO_x$ emissions, but this was not considered in the modelling, neither was possible effects on emissions due to changes in vehicle speed. Total $PM_{10}$ concentrations with the Stockholm trial were estimated to decrease by 4%-7% on four of the streets and remain unchanged on one street. For $NO_x$ the corresponding decrease is 2%-12%, being unchanged on one street. The calculated reductions are not large enough to achieve the limit values.

### 2.2.3 Conclusions from statistical analysis and dispersion modelling in Stockholm

The dispersion modelling and the statistical analysis of monitoring data in Stockholm are not comparable as they were done for different time periods and different meteorological conditions. The modelling was done assuming 15% less traffic in the CCZ and using statistical weather based on 10 years of meteorological measurements. Due to the large influence of meteorology and other factors that affect the emissions (e g other measures taken), a representative reference station with both air quality and meteorological data and a long measurement period before and after is needed in order to use monitoring data for assessing the effectiveness of a LEZ/CCS. In Stockholm, $PM_{10}$ is mainly due to road dust emissions that exhibit a strong seasonal variation that depend on road wetness conditions, and long-range transport. This makes assessments of the effectiveness of the CCS very uncertain for $PM_{10}$. $NO_x$ is mainly due to local vehicle emissions, which is likely the reason for the more robust results using $NO_x$ in the statistical analysis.

## 2.3 LEZs in Augsburg, Berlin and Munich

### 2.3.1 Statistical analysis of $PM_{10}$ data

Statistical analysis of $PM_{10}$ observations were used to evaluate the effectiveness of Low Emission Zone (LEZ) in three German cities: Augsburg, Munich and Berlin. In Munich also a second measure (truck transit ban through the city area) was implemented shortly before the LEZ became effective. The statistical modelling is based on the approach as described by Fensterer et al. (2014). In each city, the effectiveness of LEZ at different types of monitoring sites (traffic site and urban background site) was evaluated.

Overall, effects of LEZ varied with seasons and changes between monitoring stations. From the statistical model rather inconsistent and weak effects of LEZ were observed for Augsburg, while consistent and significant effects were observed for Munich and Berlin. The magnitude of the reduction was larger for Berlin, the decrease of $PM_{10}$ concentration in Berlin range between -6 and -19% depending on the monitoring site and the active stage of the LEZ.

We also observed clear seasonal differences in the statistical evaluation of the LEZ effect. The reduction of $PM_{10}$ levels in general appeared as more pronounced in summer when compared to the winter season. It is not clear if this can be attributed to a stronger LEZ effect during summer or if there are other factors that contribute to this difference.

### 2.3.2 Dispersion modelling of LEZs effects in Augsburg

An alternative and independent method of assessing the LEZ effect has been performed for the Augsburg case. $PM_{10}$, $NO_2$ and $NO_x$ urban background concentrations in Augsburg were analysed by running a dispersion model with available emission data. The local contributions to the urban background $PM_{10}$ and $NO_x/NO_2$ for this city are about 10% and 33-50% respectively of the monitored levels, whereas a regional contribution from sources outside Augsburg explains the rest for $PM_{10}$. The expected reductions of the urban background levels due to the LEZ enforcement are up to 1-2% for $PM_{10}$ and 10% for $NO_2$ by 2013. The Augsburg modelling analysis was based on two important assumptions, that also vehicles outside the relatively small LEZ area were following the LEZ requirements and that the LEZ did not affect traffic volumes inside the LEZ area.

### 2.3.3 Conclusions from statistical analyses and dispersion modelling in Augsburg

LEZ effects on air quality in Augsburg have thus been evaluated separately by both a statistical analysis of monitoring data and by running an air pollution dispersion model. The statistical modelling showed only weak effects in street $PM_{10}$ levels and could not find significant effects of LEZ in the urban background $PM_{10}$

data. The results of the dispersion modelling analysis highlight the small contribution of local traffic to $PM_{10}$ urban background levels. Year-to-year variation in regional background concentrations is larger than the expected LEZ reductions.

## 2.4 Summary of studies as reported in the literature

### 2.4.1 Studies using monitoring data

The efficacy of CSSs and or LEZs have been assessed for at least 14 cities (Augsburg, Berlin, London, Stockholm, Munich, Lisbon, Copenhagen, Milan, Rotterdam, Amsterdam, The Hague, Den Bosch, Utrecht, Tilburg) based on more or less advanced analyses of monitoring data at one or several sites in the cities. In most cases studies have used monitoring of $PM_{10}$ and $NO_2$ or $NO_x$, but some few have also analysed the effect on elemental carbon (EC), total carbon (TC = sum of elemental carbon, EC, and organic carbon, OC), absorbance and/or total particle number concentrations. The results vary considerably, especially for $PM_{10}$, which in some studies are shown to be reduced by several percent and in others not at all. As discussed above $PM_{10}$ may not be the best indicator as there are many other factors than local traffic emissions that affect its concentrations, but there may also be large uncertainties in the analyses methods used. What seems clear and logical is that the effects of a LEZ on NOx, EC or TC is larger and more robust (less uncertain) than for $PM_{10}$ due the large local traffic influence on the former parameters. The advantage of using $PM_{10}$ and $NO_2$ is that its concentration is regulated in the EU directive, which has to be achieved in cities.

### 2.4.2 Studies using dispersion modelling

Dispersion modelling has been used to assess the efficacy of LEZ/CCS in 11 cities (Stockholm, Augsburg, London, Rome, Rotterdam, Amsterdam, Copenhagen, Fredriksberg, Aarhus, Odense, and Aalborg). Different types of dispersion models have been used with different methodologies for estimating the effect on concentrations, either on some single streets in the cities or over the whole city. Calculations have been done for $NO_2$, $NO_x$, $PM_{10}$ and EC concentrations. Calculations show that the effect of a LEZ depend on the regulations in relation to existing vehicle composition and the timing of the introduction of the LEZ; the earlier the more efficient it will be. In most cases, strongest effects are seen on NOx and EC. Emission factors for EC is not readily available and likely connected with larger uncertainties compared to $NO_x$ (as EC is not regulated in the vehicle emission directives).

### 2.4.3 Use dispersion modelling or monitoring data for LEZ/CCS evaluation?

Air quality dispersion modelling is a powerful tool that can be used to estimate the potential effect on air quality of a LEZ/CCS. A prerequisite is to ascertain detailed and reliable data on changes in traffic intensities and vehicle composition, and to estimate realistic real-world emission factors representative for the local driving conditions in the city. Dispersion model calculations can also provide information on the relative importance on the concentrations of emission changes and meteorological variability, as well as spatial variability in concentrations (not only point estimates for measurement stations) and population exposure for health assessment. Due to the large influence of meteorology and other factors that affect the emissions (e g other measures taken), a representative reference station with both air quality and meteorological data and a long continuous time series with measurements before and after the introduction of the LEZ/CCS, is needed in order to use monitoring data for assessing the effectiveness of a LEZ/CCS. Ideally, both methodologies can provide valuable and complementary information on the efficacy of a LEZ/CCS for improving air quality.

## 2.5 Recommendations

In order to make a proper evaluation of LEZ or CCS on air quality it is very important to monitor the impacts on traffic intensities and vehicle composition, not only inside the zone, but also outside. Automatic number plate recognition (APNR) can be used to monitor vehicle composition and to monitor compliance of LEZ regulations. They may be more expensive in the short term than manual enforcement methods but less expensive once in operation.

Exceedances of the limit values of $PM_{10}$ and $NO_2$, are most often the main driver for introducing a LEZ. But the benefits of LEZ for the health of the population may be significant even if the reductions in $PM_{10}$ and $NO_2$ concentrations are small, since LEZ reduce exhaust combustion particulate (including black carbon) that may have the highest adverse health impacts. It is therefore recommended to complement measurement programs with pollutants that are better specific markers for vehicle exhaust emissions. In this respect soot particle concentrations (measured as BC, EC or absorbance) or particle number concentrations are better indicators than $PM_{10}$ and $NO_2$. Non-exhaust pollutants may be affected if traffic intensity or driving conditions change. For example, less congestion may reduce brake wear emissions, but may also increase mean vehicle speed which in turn affect suspension of road dust. This means that the efficiency of a LEZ depends on what pollutant that is studied. Elemental carbon (respectively Black Carbon BC) or $NO_x$ which are emitted mainly by traffic are more influenced by local road traffic exhaust and thus more affected than $PM_{10}$ or $PM_{2.5}$ to detect and demonstrate the efficacy of traffic restrictions.

There are several other measures than LEZ and CCS that have been shown to significantly improve air quality. LEZs affect only exhaust components, but CCS and other traffic access restrictions aim at reduce traffic intensities and may affect all traffic emitted pollutants.

# 3 Background

Low Emission Zones (LEZs) are areas where only vehicles that fulfil certain emission requirements are allowed to enter. If non-compliant vehicles enter they have pay a fine, or in some cases they are allowed to enter if they pay a charge. Congestion Charge Zones (CCZs) are areas to which all drivers have to pay to enter, irrespective of emission standard, even though exemptions for low emission vehicles have been applied. Both LEZs and Congestion Charge Systems (CCSs) may reduce traffic emissions and improve air quality even if they have different purposes (abate traffic emission or congestion). LEZs can be a very efficient way to mitigate air pollution in cities.

The first LEZs in Europe were set up in the Swedish cities Stockholm, Gothenburg and Malmö in 1996. Today there are more than 200 LEZs in more than 10 countries with varying degrees of restrictions (http://lowemissionzones.eu/). Cities with LEZs vary in size from big cities like London, Berlin and Milan to small towns or only some part of a highway close to the city. The motivation to introduce LEZs is of course to protect the citizens from adverse health effects and contributing to compliance with the limit values of the European Air Quality directives. Most often, exceedances of the limit values for $PM_{10}$ and $NO_2$ has been the main driver for introducing LEZs in Europe, but there are also a considerable number of cities with LEZ but no limit value exceedances (Ecorys, 2014). Emission reductions will also be beneficial for lowering the environmental and climate impact of air pollutants mainly through the reduction of $NO_x$, VOC and black carbon (BC) emissions ($NO_x$ and VOC being precursors of ozone and contributing to eutrophication and black carbon contributing to regional climate change, especially melting of ice-sheets). LEZs may also reduce noise pollution if they speed up introduction of newer vehicles that are quieter.

# 4 Objectives of the study

The objective of this study was to assess the effect on air quality of a Congestion charging system (CCS) in Stockholm and LEZs in three German cities (Augsburg, Munich and Berlin).

The analyses are based on monitoring data at street locations and urban background as well as air quality dispersion modelling using meteorological dispersion models. The analyses of monitoring data consider meteorological effects and other influencing factors in order to distinguish the CCS and LEZ effects and these analyses are compared with the atmospheric dispersion modelling based on high-resolution emission inventories for years before/without and after/with enforcements of CCS/LEZ.

The results from the cities are inter-compared and also compared with other assessments made in European cities, based on both measurements and modelling. Based on reported assessments in the literature, we discuss the importance of the methodology for demonstrating and detect the effects on air quality of a LEZ. We also discuss the difficulties encountered with attributing changes in air quality to the LEZ, advantages and disadvantages of using different methods, and pose some recommendations for future evaluations.

# 5 Effects on air quality of Congestion Tax in Stockholm

## 5.1 Description of the congestion tax zone in Stockholm

On June 16, 2004 the Swedish Parliament adopted the Congestion Charge Law (SFS 2004:629). The law made it possible to charge a congestion tax in Stockholm up to July 31, 2006. This law involved charging relatively higher prices for travel during periods of peak hours than in other periods. Before the permanent congestion charge system was in operation there was a 7 month trial during January 3 to July 31, 2006. The Stockholm Trial consisted of three parts: extended public transport, congestion tax and more park-and-ride sites in the city and the county. Then there was a period without congestion charging between August 1, 2006 and July 31, 2007. The permanent congestion tax was implemented from August 1, 2007. The permanent system has the same geographical limitations and tax levels as during the trial.

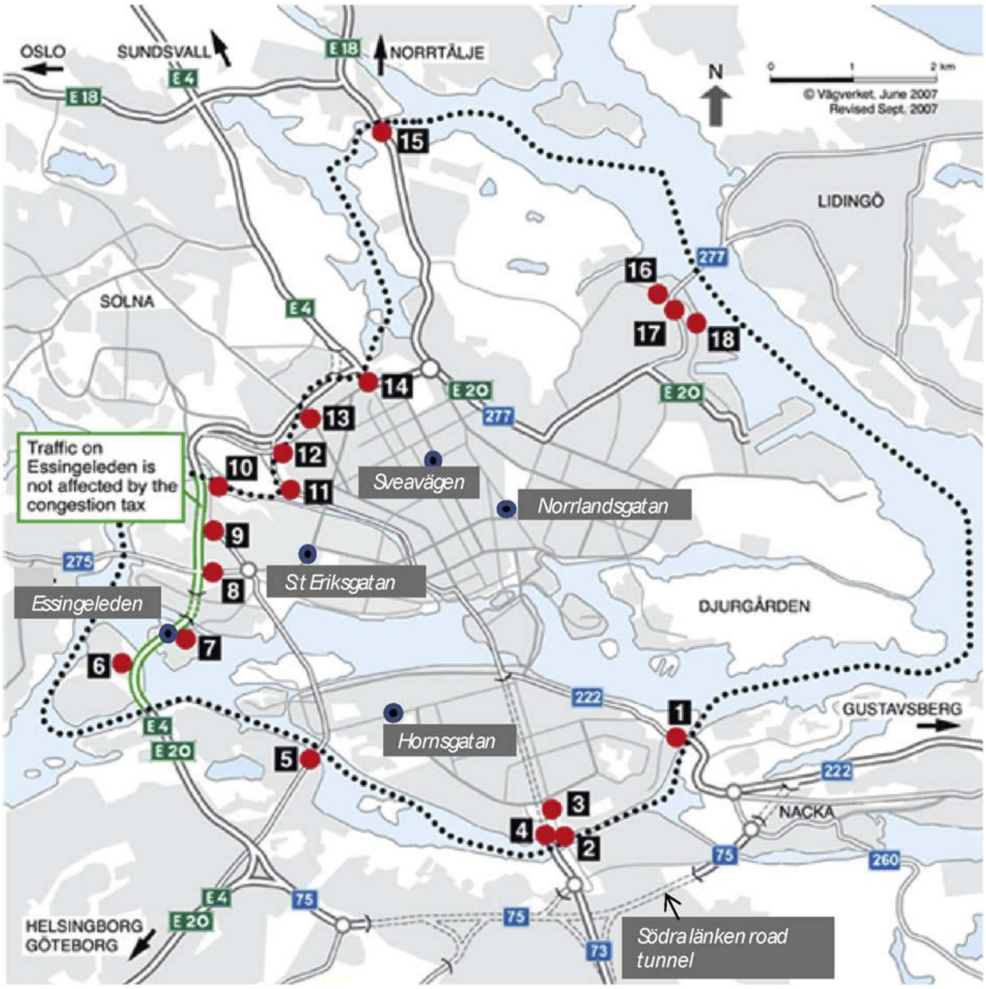



**Figure 1. Map showing the 18 toll stations of the congestion zone during the Stockholm trial (January–July, 2006). Dotted line indicates the inner city area which is reached only by passing a toll station. Black filled circles indicate sites mentioned in the text.**

Figure 1 shows the congestion charge zone with the 18 toll stations and the location of the air quality monitoring stations. Congestion tax is charged for travelling into and out of the inner city of Stockholm between 6.30 am and 6.30 pm. The fee varies with different time intervals between 10 kronor and 20 kronor, with a maximum charge of 60 kronor per day per vehicle. Evenings, nights, Saturdays, public holidays and days before public holidays are free of charge, and certain types of vehicles are exempt. The congestion tax does not apply to traffic on the bypass Essingeleden, nor for journeys to and from Lidingö Island which pass through the inner city within a window of 30 minutes. Parts of the expansion of public transport that was a part of the Stockholm Trial have also been made permanent. The main remaining differences are:

- The congestion tax is deductible for business and work-related journeys
- The congestion tax is invoiced monthly
- The exemption for alternative fuel vehicles is discontinued on 1 January 2012

- Alternative fuel vehicles registered after 1 January 2009 are not exempt
- The exemption for taxis and transport for the disabled is discontinued
- The month of July is free of congestion tax

## 5.2 Evaluation of the CCS in Stockholm based on air quality dispersion modelling

The impact on air quality and health of a road pricing system (Congestion Charge System, CCS) in Stockholm has been assessed earlier using air quality dispersion modelling (Johansson et al., 2009). This assessment was made for 2006 when the system was introduced as a trial for 7 months. In this report, we shortly summarize the method and results of this modelling. The modelled concentrations are annual mean values, assuming that the effects of the 7 month trial on traffic would represent one full year.

### 5.2.1 Methods

#### 5.2.1.1 Modelling

The annual mean $PM_{10}$ and $NO_x$ concentrations and exposures due to local road traffic emissions were calculated using a diagnostic wind model and a Gaussian dispersion model. Meteorological conditions were based on a climatology that was created from 10 years of meteorological measurements (15 minute averages) in a 50 meters high mast located in the southern part of Stockholm. This climatology is assumed to represent typical meteorological conditions, i.e. not the meteorological conditions for a specific year. The dispersion calculations were performed for Greater Stockholm (1.44 million inhabitants, 35 x 35 km) on a 100 meter resolution (122 500 receptor points).

#### 5.2.1.2 Emissions

Road traffic was quantified in terms of traffic flow by counting vehicles and by calculating road use, i.e. the number of vehicle kilometres travelled in the area. Congestion was quantified in terms of journey times obtained from floating car measurements or from traffic cameras. Emissions from road traffic are described with emission factors for passenger cars (petrol and diesel), light commercial vehicles, heavy goods vehicles. Emission factors for $NO_x$ were obtained from the Swedish EVA model of the Swedish Road Administration and refer to year 2006.

### 5.2.2 Results of modelling impact of Stockholm CCS

#### 5.2.2.1 Impact of CCS on emissions

Compared with the situation with no CCS, $NO_x$ emissions in the Greater Stockholm area would decrease by 55±16 tonnes as result of reduced traffic. Most of this reduction occurs in Stockholm's inner city; 45±13 tonnes. For $PM_{10}$, the corresponding reduction would be 30±9 tonnes, of which about two thirds would be the result of reductions in emissions in the inner city. Both exhaust and non-exhaust particles would have decreased. Increased $NO_x$ emission from busses compensate for some of the reduction in emissions from passenger cars. This is the reason for $NO_x$ emissions to be reduced by only 8.5% in the inner city, while total road use is reduced by 15%. For the other substances the emissions decrease approximately to the same extent (in relative terms) as the road use. Reduced traffic congestion further lowers the emissions by 1 % seen across the entire 24-hour period, and by 2-3 % at periods of peak traffic, but this is not taken into account in the modelling.

#### 5.2.2.2 Impact of CCS on concentrations

Figure 2 shows the geographic variations of the annual mean reduction in $NO_x$ and $PM_{10}$ concentrations due to the reduced traffic. Note that these concentrations represent roof top, not street level (the model cannot resolve individual streets and building effects). Largest reductions are seen in the city centre, where concentrations are estimated to fall by up to 2 μg/m$^3$ for both $NO_x$ and $PM_{10}$. The greatest improvements are found along the city centre and south of the city centre in connection to a road tunnel bypass (Södra länken). Concentrations of $NO_x$ and $PM_{10}$ increase in an area around the toll-free Essingeleden and near the portals of the road tunnel, due to the increase in traffic emissions.

Table 1 gives some examples of estimated effects of CCS on total annual mean $NO_x$, $NO_2$ and $PM_{10}$ concentrations along some densely trafficked streets. Total $PM_{10}$ concentrations with the Stockholm trial are estimated to decrease by 4%-7% on four of the streets and remain unchanged on one street. For $NO_x$ the corresponding decrease is 2%-12%, being unchanged on one street. The calculated reductions are not large enough to achieve the limit values.

More details, including health benefit calculations based on mean population exposure estimates, are given in Johansson et al. (2009).

**Table 1. Estimated total concentrations and percentage changes in levels along some streets in the inner City of Stockholm due to the Stockholm CCS (annual mean values). The locations of streets are shown in Figure 1.**

| **Steet in City center** | **Number of vehicles per day (Weekdays)** | $NO_x$ [1)] | $NO_2$ [2)] | $PM_{10}$ [3)] |
|---|---|---|---|---|
| Hornsgatan | 35 000<br>-8% | 103 µg/m³<br>- 8% | 77 µg/m³<br>-3% | 78 µg/m³<br>-5% |
| Sveavägen | 30 000<br>-6% | 76 µg/m³<br>-2% | 68 µg/m³<br>-1% | 62 µg/m³<br>-4% |
| Norrlandsgatan | 10 000<br>-12% | 77 µg/m³<br>-11% | 68 µg/m³<br>-5% | 62 µg/m³<br>-7% |
| Valhallavägen | 38 000<br>-14% | 35 µg/m³<br>-12% | 47 µg/m³<br>-7% | 58 µg/m³<br>-7% |
| S:t Eriksgatan | 35 000<br>+5% | 53 µg/m³<br>unchanged | 58 µg/m³<br>unchanged | 57 µg/m³<br>Unchanged |
| Essingeleden | 140 000<br>+3% | 54 µg/m³<br>+ 2% | 58 µg/m³<br>+ 1% | 67 µg/m³<br>+1% |

[1)] *Annual mean value.*

[2)] *98th percentile of daily mean values. The Swedish limit value is 60* µg/m³. *(There is no EU limit value for daily mean* $NO_2$*)*

[3)] *90th percentile of daily mean values. EU and Swedish limit value is 50* µg/m³.

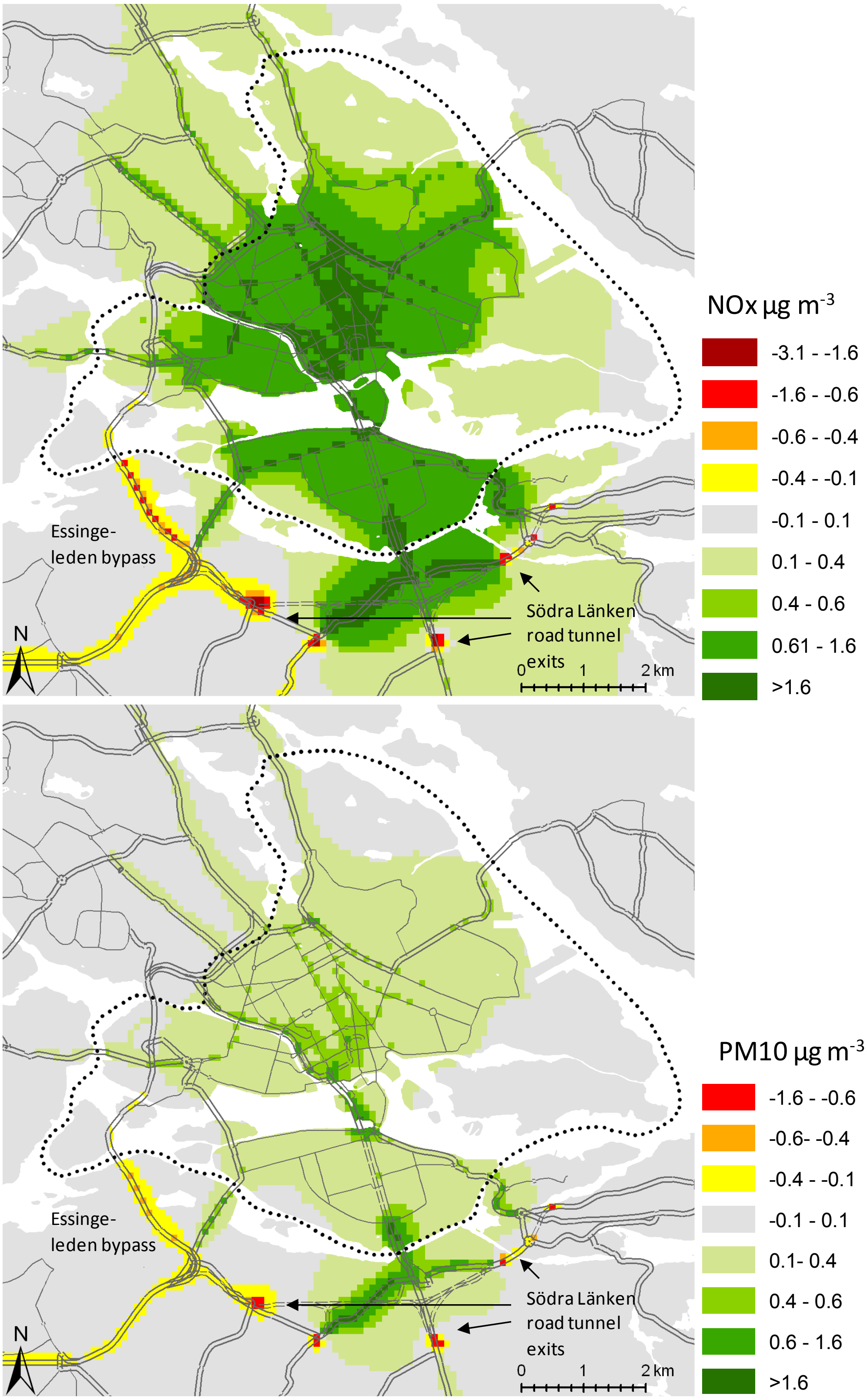



**Figure 2. Difference in annual mean concentrations of $NO_x$ and $PM_{10}$ with the Stockholm Trial compared to a situation without the Trial (with the same meteorology and other emissions than road traffic). Within the green areas the levels have fallen, within yellow to red areas there is an increase in levels. In the inner city changes refer to rooftop height (not street canyon).**

## 5.3 Statistical analysis of Stockholm's CCS based on monitoring data

Here we present statistical analyses based on ambient monitoring of air quality and meteorology. The analyses were made by Johannes Deggendorfer and Solange Ducrocq, students at the Institute for Statistics, Ludwig-Maximilians-University in Munich under the supervision of Veronica Fensterer and Professor Helmut Küchenhoff (Ducrocq and Deggendorfer, 2014). Below is a summary of their report.

### 5.3.1 Method

The statistical analysis of the effects of the CCS in Stockholm, Sweden is based on a semiparametric statistical model with first-order autoregressive errors. This model has also successfully been applied to Munich and is described in more detail by Fensterer et al. (2014). Hourly mean values of $PM_{10}$ and $NO_x$ and meteorological parameters from 2005 to 2008 were used. There were two 6 month periods **with** congestion tax (January 3rd to June 30th, 2006; Second: January 3rd to June 30th, 2008) and two 6 month periods **without** tax (January 3rd to June 30th, 2005; Second: January 3rd to June 30th, 2007).

Figure 3 shows the location of the air pollution monitoring stations. The station at Essingeleden was used as reference site. This is also located inside the tax cordon, but the highway bypass Essingeleden was exempted from the congestion tax. There are around 140 000 vehicles per day on Essingelden and the measurement station is located just 3 m from the road with the air intake at 3 m height.

Hornsgatan, Sveavägen and Norrlandsgatan have measurements at kerbsides in street canyons, at ca 3 m height and 2 m from the façade. At Hornsgatan and Sveavägen $NO_x$ was measured at kerbsides on both sides of the street and at Norrlandsgatan only on one side. $PM_{10}$ was measured at one side of the streets.

NOx was also measured on the roof-top of Hornsgatan, Sveavägen and Torkel Knutssonsgatan and $PM_{10}$ only on the roof of Torkel Knutssonsgatan. Air intakes are ca 2 m from the roof at ca 27 m above the street surfaces.

Data on atmospheric pressure, precipitation and relative humidity are from a meteorological 10 m high mast on the roof-top at Torkel Knutssonsgatan. Data on ambient temperature, temperature difference (between 20 and 5 m), wind speed, wind direction, and global radiation are from a 50 m high mast located in Högdalen a few km's south of the city center.

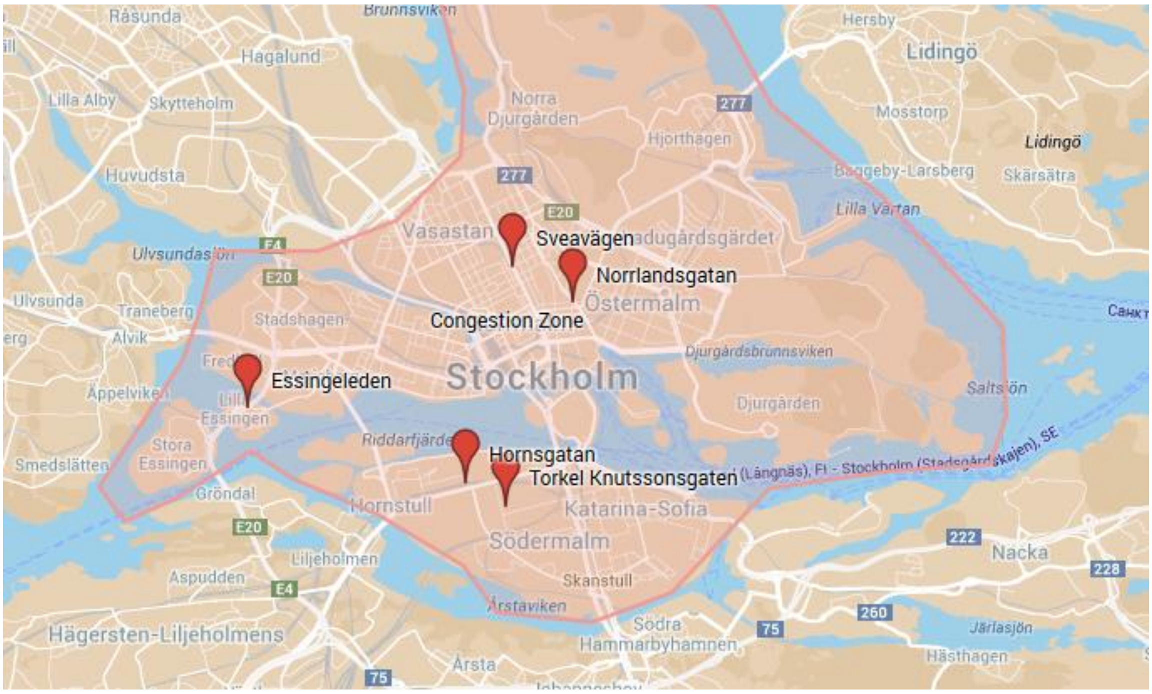



**Figure 3. Locations of air pollution monitoring stations providing data for the statistical modelling. The reference site is also located inside the tax cordon (red line), but the highway bypass (Essingeleden) was exempted from congestion tax.**

Variable selection was done using Generelized Cross Validation Score (GSV-score). The smaller the GCV, the better the model. For $NO_x$ the final model is formulated as:

$$\log(NO_{xi}) = \beta_0 + f(\log(NO_{xref})) + \beta_1 Holidays + \beta_W I_W + \beta S_W I_{SW} + \beta_{WW} I_{WW} + f_S(Hour2) I_S + f_W(Hour2) I_W + f_{SW}(Hour2) I_{SW} + f_{WW}(Hour2) I_{WW} + f(Windspeed) + f(Winddirection) + f(Temperature) + f(Temp\_difference) + \varepsilon_i$$

And for $PM_{10}$:

$$\log(PM_{10i}) = \beta_0 + f(\log(PM0_{ref})) + \beta_1 Holidays + \beta_W I_W + bS_W I_{SW} + \beta_{WW} I_{WW} + f_S(Hour2) I_S + f_W(Hour2) I_W + f_{SW}(Hour2) I_{SW} + f_{WW}(Hour2) I_{WW} + f(Windspeed) + f(Winddirection) + f(Temperature) + f(Temp\_difference) + f(Precipitation) + \varepsilon_i$$

With:

*W : Winter without congestion tax*

*S: Summer without congestion tax*

*WW: Winter with congestion tax*

*SW: Summer with congestion tax*

*I: Indicator of the interaction between newseason and measure*

*Hour2: Hour variable over a complete week*

For weekly effects, a new hour variable for the whole week, therewith the effects of the different days of the week could be illustrated. January the 1st was removed, because of the biased values, due to New Year's Eve. For the $PM_{10}$ model the following variables were selected: public holidays, temperature, temperature difference and precipitation. For the $NO_x$ model we selected the same variables, except for precipitation. For more detailed description, see Ducrocq and Deggendorfer (2014).

### 5.3.2 Results from the statistical analyses

For most monitoring sites all the variables have a significant influence on $PM_{10}$ and $NO_x$. There are some cases where the variable is not significant, for example, the variable "holidays" is non-significant, at Norrlandsgatan for $PM_{10}$. The estimated parameter β for this variable is almost always negative, which means that with a holiday, there is less pollution. Figure 4 shows boxplots of $NO_x$ concentrations for all periods with and without tax. There is a systematic decreasing trend in concentrations, irrespective of tax or not. This is due to the changing vehicle composition, lower emissions from new cars compared to old. Roof-top concentrations (Horns 3, Svea 3 and Tork Kn) are much lower than those at kerbside.

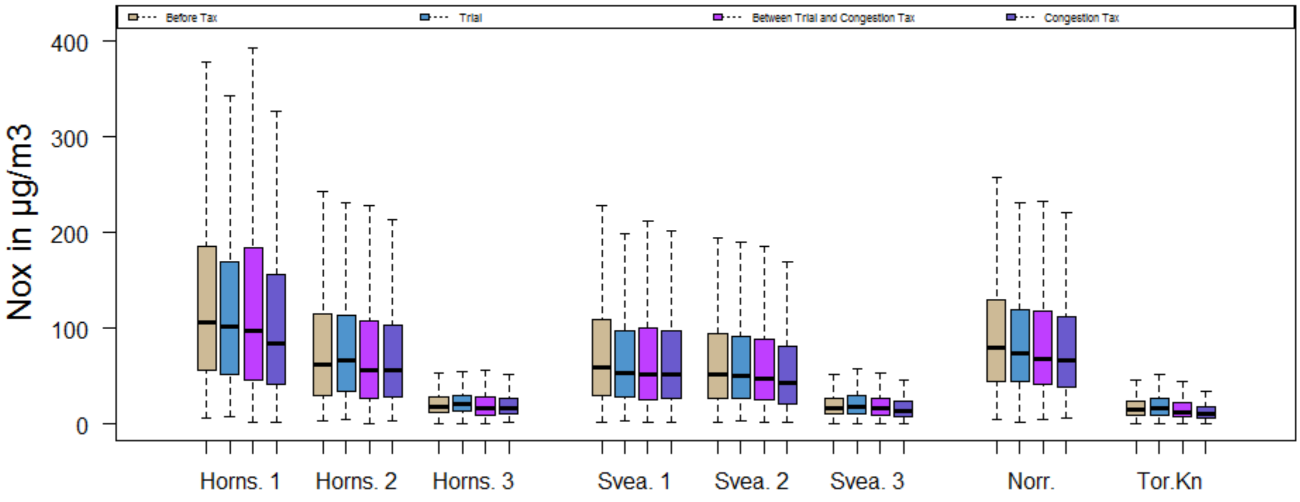



**Figure 4. NOx concentrations (maximum, minimum, median and quartiles) during the different periods with and without tax.**

Figure 5 shows the temporal variation of the $PM_{10}$ concentrations during all periods. Roof-top concentrations are much lower than those at kerbside and there are higher values during spring time due to suspension of road dust from dry road surfaces. The spring time peaks varies depending on the wetness of the roads during this period.

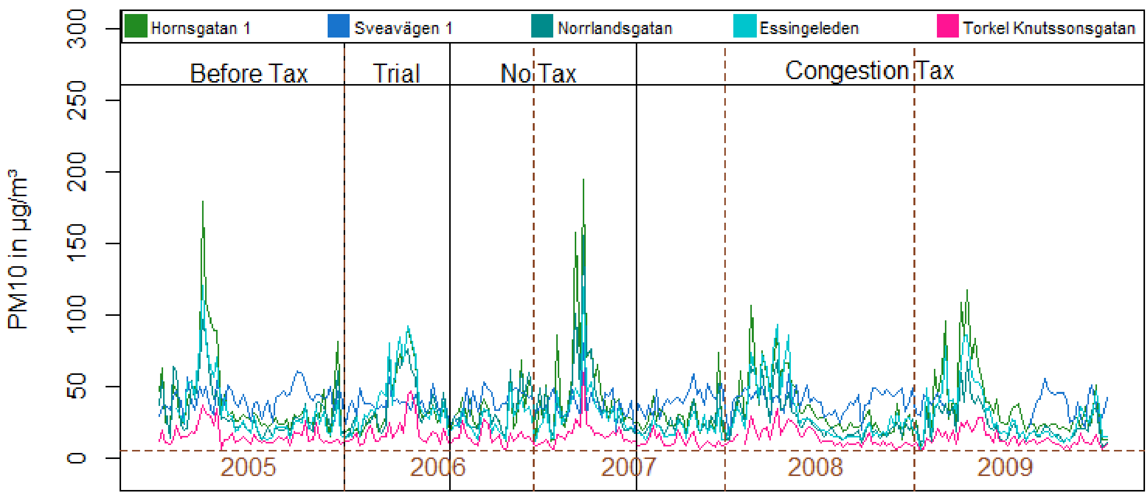



**Figure 5. Weekly mean $PM_{10}$ concentrations during the whole period. Light red line shows concentrations at roof-top stations and the other lines are concentrations from kerbside stations.**

As shown in Figure 6 there is no clear trend in $PM_{10}$ concentrations. As only a small fraction of $PM_{10}$ is due to exhaust PM, the renewal of the vehicle fleet, potentially leading to lower emissions, has little effect on $PM_{10}$ concentrations. Rod dust suspension is the main local source of $PM_{10}$ and it is dependent on meteorological conditions, traffic intensities, vehicle speed and the share of studded winter tires. Congestion charging may affect traffic intensities and vehicle speed.

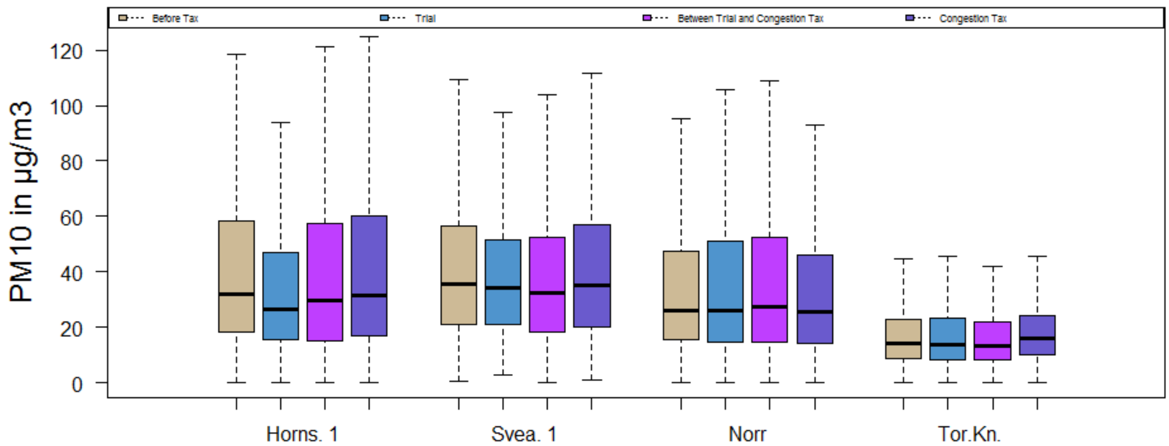



**Figure 6. $PM_{10}$ concentrations (maximum, minimum, median and quartiles) during the different periods with and without tax.**

Figure 7 shows that for periods with congestion tax $NO_x$ concentrations were reduced during daytime on weekdays, but not during nighttime and not weekends. This is consistent with the fact that nights and weekends are free of tax. This also indicates that the model is not influenced by the overall decreasing trends in concentrations during the study period (the periods with congestion tax represent more recent data that would have had lower concentrations due to newer vehicle fleet compared to periods without congestion tax, even if the congestion tax would not have been implemented).

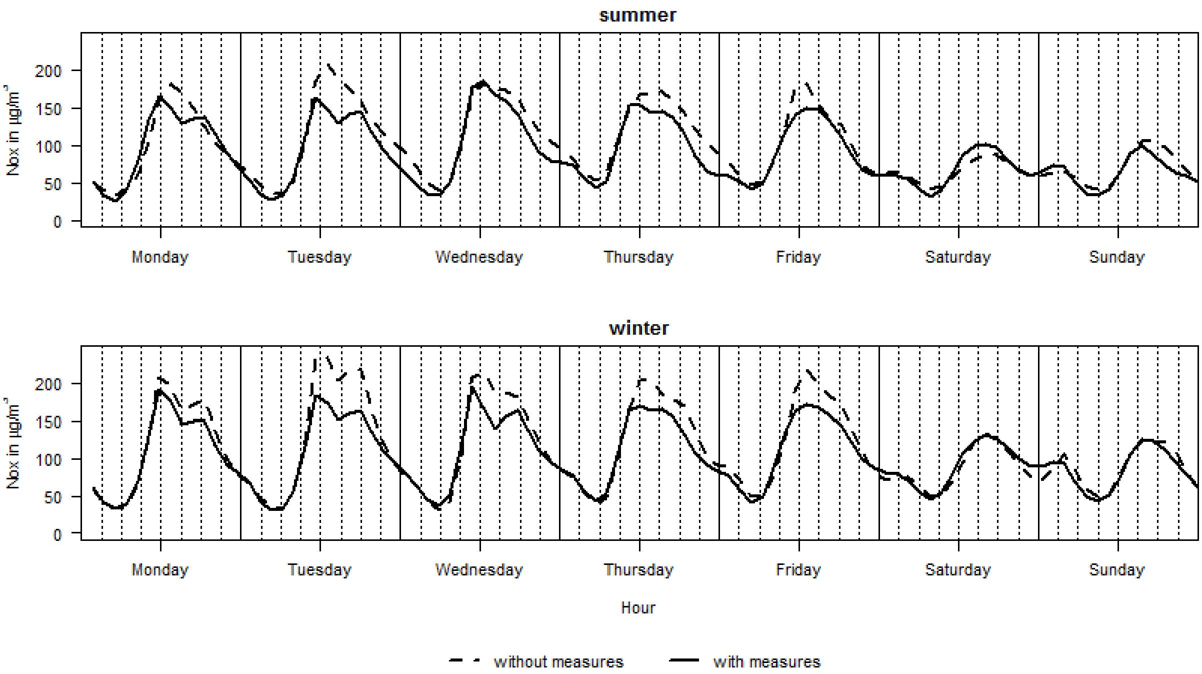



**Figure 7. Mean model calculated $NO_x$ with and without congestion tax for Hornsgatan, kerbside on north side on all Mondays to Sundays for summer and winter.**

Figure 8 shows the daily variations on weekdays and weekends for $PM_{10}$ with and without congestion tax, for summer and winter. There is a very large difference in diurnal variation during summer and winter. The main reason is the high $PM_{10}$ concentrations during the spring as indicted in Figure 5.

The tendency of higher $PM_{10}$ concentrations without tax in winter may partly be due to streets being dryer during the periods without tax. So this may have very little or nothing to do with the congestion tax.

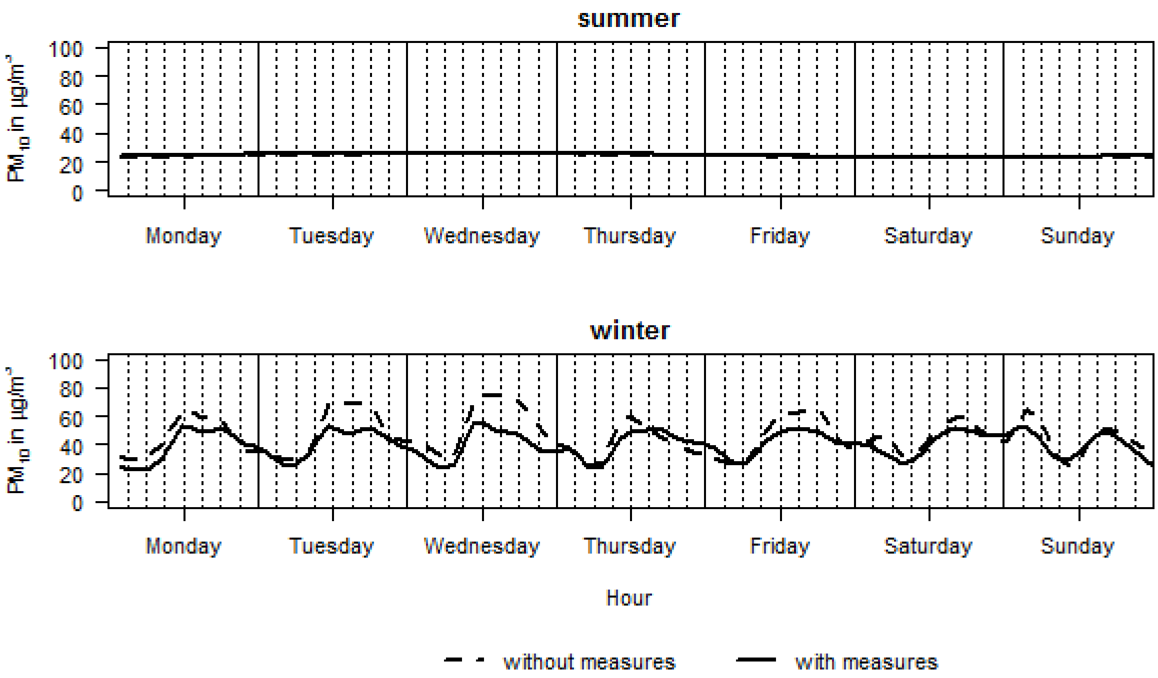



**Figure 8. Mean model calculated $PM_{10}$ with and without congestion tax for Hornsgatan, kerbside on north side on all Mondays to Sundays for summer and winter.**

Table 2 shows the calculated difference in $NO_x$ and $PM_{10}$ concentrations for periods with congestion tax compared to periods without congestion tax. The effect is calculated separately for summer and winter as well as for a whole year.

For $NO_x$ all measurement stations, except the roof-top site at Hornsgatan shows lower concentrations during congestion tax periods as compared to non-tax periods. The reduction ranges from -0.1% to -9.2%. Looking at summer only, all stations show reduced concentrations of $NO_x$ (-6.1% to -18%). For winter, reductions are smaller and one of the street stations (Hornsgatan, southern side) and one roof-top station (Hornsgatan) even show increased $NO_x$ concentrations.

For $PM_{10}$ results are less clear, but there are fewer sites with measurements. Two (of three) street sites show reduced concentrations by -8.4% and -10.3%, but one show a small increase (+0.6%). The roof top station show increased concentrations (+15.5%). The $PM_{10}$ concentrations in Stockholm are mainly due to coarse particles and strongly dependent on the weather, especially the road conditions. Spring time periods with frequent wet roads show lower concentrations than periods with dry roads. This is due to the suppression of road dust suspension during periods with wet roads. The model includes precipitation as one parameter, but it is known that this is not so good indicator of road wetness.

**Table 2. Percentage difference in $NO_x$ and $PM_{10}$ concentrations at different sites for periods with and without congestion tax.**

| | $NO_x$ | | | $PM_{10}$ | | |
|---|---|---|---|---|---|---|
| Site/Season | Winter | Summer | Whole year | Winter | Summer | Whole year |
| Hornsgatan street Northern side of road | -5.3% (-9.4%,-1.2%) | -10.2% (-16.1%,-3.8%) | -7% (-10.3%,-3.5%) <0.001* | -11.9% (-19.2%,-4.7%) | +2,7% (-9.9%,+17.1%) | -8.4% (-14.9%,-1.3%) 0.021* |
| Hornsgatan street Southern side of road | +8.3% (+2.3%,+14.4%) | -18.1% (-25%,-10.5%) | -0.1% (-4.8%,+4.8%) 0.969* | | | |
| Hornsgatan roof-top | +12.4% (+5%,+19.7%) | -7.9% (-16.9%,+2.2%) | +6% (+0.2%,+12.1%) 0.041* | | | |
| Sveavägen street Western side of road | -4.2% (-7.2%,1.1%) | -6.1% (-10.7%,1.3%) | -4.9% (-7.4%,-2.3%) <0.001* | +1.9% (-1.9%,+5.7%) | -2% (-7.7%,+4%) | +0.6% (-2.5%,+3.9%) 0.688* |
| Sveavägen street Eastern side of road | -5.6% (-9.1%,-2.1%) | -14.7% (-19.6%,-9.6%) | -9.2% (-11.1%,-5.3%) <0.001* | | | |
| Sveavägen Roof-top | -3.4% (-9.3%,2.4%) | -15.2% (-22.9%,-6.8%) | -6.9% (-11.5%,-2.0%) <0.001* | | | |
| Norrlandsgatan, street, Western side of road | -6.7% (-9.7%,-3.8%) | -7.2% (-11.8%,-2.3%) | -7.1% (-9.6%,-4.6%) <0.001* | -14.7% (-20.5%,-8.9%) | +4.7% (-6.3%,+16%) | -10.3% (-15.6%,-4.7%) <0.001* |
| Torkel Knutssonsgatan, roof-top | -5.0% (-11.1%,+1.1%) | -7.8% (-16.8%,+2.1%) | -5.7% (-10.7%,-0.4%) 0.037* | +19.4% (+7.1%,+31.7%) | +6.9% (-9.1%,+25.9%) | +15.5% (+5.8%,+26%) 0.001* |

* p-value

It is implicitly assumed that the differences in concentrations between periods with congestion tax and those without charging at the reference site are due to factors that are not related to the charging (e g changes in vehicle composition leading to lower emissions). But since the charging lead to some increase (few percent) in traffic intensity on the highway at the reference site as it was exempted from tax, the effects of the tax on the concentrations inside the tax cordon is likely overestimated.

### 5.3.3 Conclusions

The statistical analyses using monitoring data of $NO_x$ from several sites in central Stockholm indicates that the congestion tax system have contributed to reduced concentrations of $NO_x$ at kerbside sites. For roof-top locations results are not consistent. For $PM_{10}$ the picture is less clear, likely due to the dominating influence of road dust suspension, which in turn is mostly dependent on weather conditions during the winter and spring period (Norman et al., 2006).

# 6 Effects on air quality of low emission zones in Germany

## 6.1 Background

In Germany a LEZ is a defined area (mostly located in the city centre) where vehicles have to meet emission standards when entering it. Vehicles in Germany are identified by windscreen badges that come in a coloured code which is directly linked to the corresponding stages of European emission standards.

Table 3 shows the regulations of sticker assignment for LEZs in Germany. Gasoline-powered vehicles equipped with a catalytic converter are principally Euro 4 and will be entitled a green badge. Stickers for diesel-powered vehicles are dependent on the European emission standards (red for Euro 2, yellow for Euro 3 and Green for Euro 4). The LEZs may only be entered by vehicles with the emission category permitted for that zone. In principle, the German LEZs could be operated in three different stages: in the first stage all vehicles with emission sticker (red, yellow or green) are allowed to enter the LEZ, in the second stage only vehicles with yellow and green stickers are allowed to enter the LEZ and finally in the third stage only vehicles with green stickers are allowed to enter the LEZ.

Table 4 compares the LEZs implemented in Augsburg, Munich and Berlin. In general, the LEZs differ largely in size and implementation time from city to city. The details of LEZ for each city will be discussed in the following.

**Table 3. German regulations of sticker assignments for LEZs.**

| Emission group | 1 | 2 | 3 | 4 |
|---|---|---|---|---|
| | No sticker | Red sticker | Yellow sticker | Green sticker |
| | |  |  |  |
| Gasoline powered vehicles | Euro 1 or less (vehicles which do not meet group 4 criteria – mostly without a catalytic converter) | Sticker is not issued | Sticker is not issued | Euro 1 and higher (mostly vehicles equipped with a catalytic converter) |
| Diesel powered vehicles | Euro 1* or less | Euro 2* or Euro 1 with catalytic diesel particulate filters | Euro 3* or Euro 2 with catalytic diesel particulate filters | Euro 4* and higher or Euro 3 with catalytic diesel particulate filters or better |

** for more details regarding the emission standards Euro1, Euro2, Euro3 and Euro4 please refer to [http://www.dieselnet.com/standards/eu/ld.php.](http://www.dieselnet.com/standards/eu/ld.php)*

**Table 4. Comparison of Low Emission Zone in Augsburg, Munich and Berlin.**

| | Augsburg | Munich | Berlin |
|---|---|---|---|
| LEZ Area | ~ 6 km² (3%) | ~ 44 km² (14%) | ~ 88 km² (10%) |
| Population within LEZ | ~ 20.000 (7%) | ~ 420.000 (32%) | ~ 1.000.000 (29%) |
| Heavy traffic ban, started | cancelled | Feb. 01, 2008 | No plans |
| LEZ Stage 1 started | Jul. 01, 2009 | Oct. 01, 2008 | Jan. 01, 2008 |
| LEZ Stage 2 started | Jan. 01, 2011 | Oct. 01, 2010 | - |
| LEZ Stage 3 started | postponed | Oct. 01, 2012 | Jan. 01, 2010 |

### 6.1.1 Low Emission Zone in Augsburg

The LEZ in Augsburg covers in general the inner city area and has an area of 5.7 km² (4% of the total city area). A map of the Augsburg LEZ is shown in Figure 9. The first stage of the Low Emission Zone in Augsburg was implemented on July 1, 2009, and from January 1, 2011, the second stage of the LEZ came into force. In March, 2012, a massive renovating construction began in the city center and thus the third stage of LEZ was postponed. A heavy traffic ban plan was proposed in Augsburg but it was cancelled later.

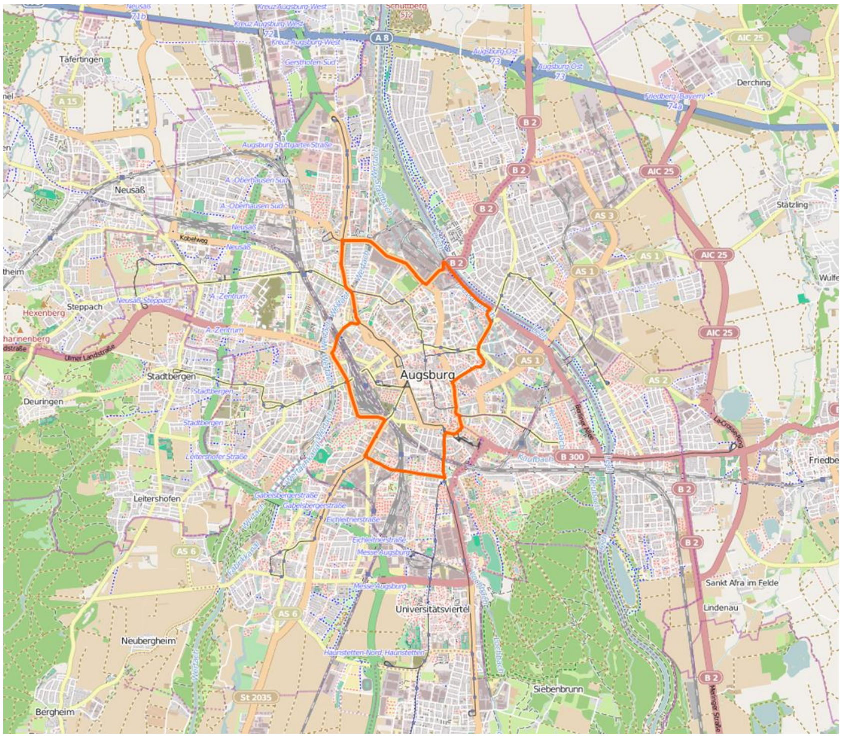



**Figure 9. Map of the LEZ in Augsburg, Germany (the border of LEZ is in red) http://www.openstreetmap.org/relation/374352#map=13/48.3690/10.8953)**

### 6.1.2 Low Emission Zone in Munich

Munich had a population of approximately 1.41 million in 2011 in an area of 310.4 km² and approximately 700 000 cars were registered (Statistisches Amt München, 2012). Figure 10 shows the map of LEZ in Munich. The LEZ in Munich covers 44 km² which accounts for 14% of the whole city area. About 32 % of the city population is living in this area. The first stage of LEZ was effective on October 1, 2008. The second stage and third stage came into force on October 1, 2010 and October 1, 2012, respectively.

Besides LEZ, there was a second measure controlling the traffic in Munich. A transit ban for heavy-duty vehicles heavier than 3.5 tons within the city area came into force on February 1, 2008. The transit ban prohibited the trucks, whose final destination is not Munich, from entering the city centre. The trucks are required to use the highway A99 which is near the outer boundary of Munch (shown in Figure 10) to bypass the city area. The heavy traffic was in effect before the introduction of LEZ, thus the analysed effects on $PM_{10}$ levels will be a combination of both control measures.

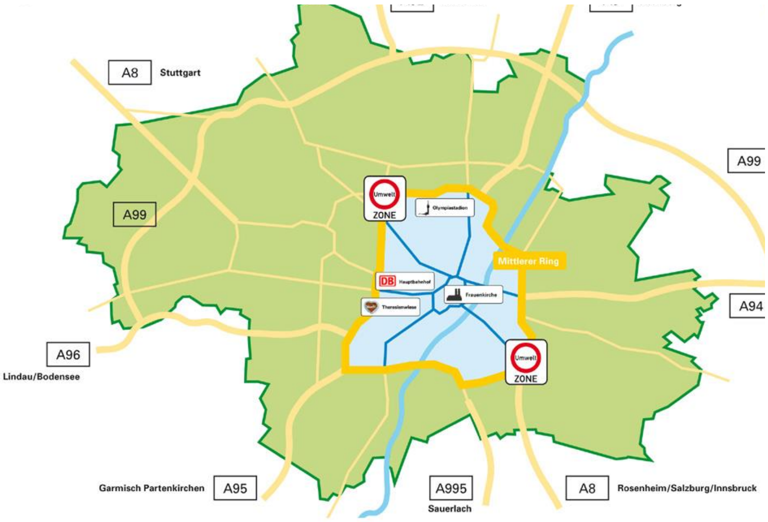



Figure 10. Map of LEZ in Munich (www.geoinfo-muenchen.de). The LEZ is surrounded by yellow line and in blue.

### 6.1.3 Low Emission Zone in Berlin

Figure 11 shows the map of LEZ in Berlin. In Berlin, the LEZ covers an area of 88 km², accounting for 10% of the total city area. 29% of the population is living within the LEZ. The first stage of LEZ was introduced on January 1, 2008. Two years later on January 1, 2010, the third stage of LEZ came into force right after the first stage. The second stage of LEZ was skipped in Berlin.


Figure 11. Map of LEZ in Berlin. The LEZ is surrounded by green line in the city centre (http://www.berlin.de/umwelt/aufgaben/verkehr-umweltzone.html).

## 6.2 Statistical Method

A direct comparison of PM concentrations before and after LEZ, or between cities with and without LEZ is difficult, if not impossible, in order to assess the effectiveness of LEZ. A lot of factors other than LEZ will affect the PM concentrations. These factors include meteorology, variations in emissions and long range transport of aerosol. Thus advanced statistical models should be used to remove these confounders, which will have an undesirable impact on PM concentration. In the project, we used a generalized additive mixed model (GAMM) to assess the effectiveness of LEZ. This model was recently used for estimation of LEZ effects in Munich (Fensterer et al., 2014).

It is a semiparametric model with first-order autoregressive errors and it is in general described by the following form (1):

$$\begin{aligned}\log(PM10_{\mathrm{X}}) = \beta_0 + \beta_1 \log(PM10_{\mathrm{ref}}) + \beta_{\mathrm{SM}} \cdot \mathrm{I}_{\mathrm{SM}} + \beta_{\mathrm{W}} \cdot \mathrm{I}_{\mathrm{W}} + \beta_{\mathrm{WM}} \cdot \mathrm{I}_{\mathrm{WM}} + \\ f_{\mathrm{S}}(hour) \cdot \mathrm{I}_{\mathrm{S}} + f_{\mathrm{SM}}(hour) \cdot \mathrm{I}_{\mathrm{SM}} + f_{\mathrm{W}}(hour) \cdot \mathrm{I}_{\mathrm{W}} + f_{\mathrm{WM}}(hour) \cdot \mathrm{I}_{\mathrm{WM}} + \\ f_{\mathrm{wd}}(wind\ direction) + \beta_2\, public\ holiday + \varepsilon.\end{aligned} \tag{1}$$

The outcome variable is the logarithmically transformed $PM_{10}$ levels at the monitoring site under study, which is denoted by $\log(PM_{10X})$. The effects of the corresponding covariates are denoted with β. $I_S$, $I_W$, $I_{SM}$, $I_{WM}$ denote the indicator function for "summer without measures", "winter without measures", "summer with measures" and "winter with measures", respectively. The winter season was defined from October to March, the summer season from April to September. The overall effect of the measures was examined for each of the two seasons with a test on the hypotheses whether the effect coefficients for winter and summer differ between the periods with and without measures.

The confounder variables were selected by a priori considerations. The logarithmically transformed $PM_{10}$ levels at the reference station ($\log(PM_{10ref})$) was used as confounder variable for changes in $PM_{10}$ concentrations which are not affected by LEZ. $PM_{10}$ levels of the reference station reflect the changes in $PM_{10}$ concentrations owing to changes of regional $PM_{10}$ levels and long term temporal trends in $PM_{10}$ levels. Furthermore, adjusting for the $PM_{10}$ measurements at the reference station prevents from "regression to the mean" (Barnett et al., 2005), improves the power of the model in comparison to the analysis of differences (Vickers et al., 2001) and allows flexible and simple adjustments for other confounders.

Temperature and precipitation were not included as confounder into the model, as the short-term changes of $PM_{10}$ levels driven by meteorological conditions are well represented by the $PM_{10}$ levels at the reference site (already included as confounder variable).

Since wind direction has a local effect, which differs from the effect at the reference station, the variable *wind direction* was included in the model as a smooth, cyclic effect based on P-splines with maximum four degrees of freedom of the wind direction ($f_{wd}(wind\ direction)$). Also an indicator for public holidays (*public holiday*) was included.

In addition, the model was adjusted for deterministic seasonal components, similar to Li et al. (1999). The effect of the measures (M) was analysed separately for summer (S) and winter (W) to allow for seasonal variability. Daily and daytime-specific deviations were modelled with an hourly-resolved weekly season-specific trend ($f_S(hour) \cdot I_S, f_{SM}(hour) \cdot I_{SM}, f_W(hour) \cdot I_W, f_{WM}(hour) \cdot I_{WM.}$).

Percentage changes of $PM_{10}$ levels were modelled through logarithmic concentration levels (Li et al., 1999). In particular, it was suggested that the measures, the public holidays and the seasons yielded to percentage effects on the $PM_{10}$ concentration.

Note, that the usage of the semiparametric model lessens the problem of scale. Furthermore, the highly skewed distribution of the $PM_{10}$ mass concentration was another reason for using the logarithmic transformation. Since the measurements are not independent, an autoregressive process of order 1 is simultaneously modelled for the error term ε.

Day-specific effects were investigated using an appropriate linear combination of the effect coefficients, representing the area between the smooth effect with and without measures. The inference for the day-specific effects was based on the asymptotic normality of linear combinations.

Statistical calculations were conducted using R (http://www.R-project.org), version 2.15.3; semiparametric models were estimated with the package "mgcv" (Wood 2006).

## 6.3 Results of the statistical analysis of $PM_{10}$ observation data

### 6.3.1 LEZ effects on $PM_{10}$ in Augsburg

#### 6.3.1.1 Measurement sites and study period in Augsburg

For the analysis we used $PM_{10}$ data from four monitoring sites which are operated by the Bavarian Environment Agency (Bayerisches Landesamt für Umwelt) as part of the Bavarian Monitoring System for Air Quality (Lufthygienisches Landesüberwachungssystem Bayern: LÜB). Figure 12 shows the locations of the measurement sites in Augsburg, Germany.

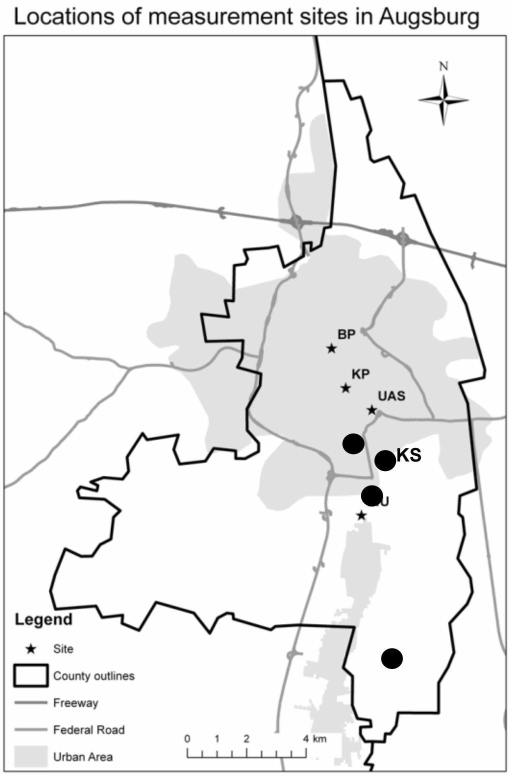



**Figure 12. Locations of the measurement sites in Augsburg (LfU site, KP: Königsplatz, BP: Bourgesplatz, KS: Karlstrasse).**

Königsplatz site is located in the city centre next to streets with high traffic density (30500 cars per day in year 2000) and is considered as a traffic site. In addition, all tram lines in Augsburg pass through KP site. Karlstrasse site (KS) is directly located at the kerbside of Karlstrasse, and is considered as a traffic site. The monitoring site at Bourgesplatz (BP) is located in a small park and is about 1 kilometer north of KP site. As a major road is located about 70 meters north of BP site, this site is considered as traffic influenced urban background site. Another official air quality monitoring station (LfU) is located on the premises of Bavarian Environment Agency (LfU), approximately 4 kilometers south of the city centre. KP, KS, and BP sites are located within in the LEZ, while LfU site lies outside the LEZ.

**Three time periods were chosen depending on the LEZ stages in Augsburg (**

Figure 13).

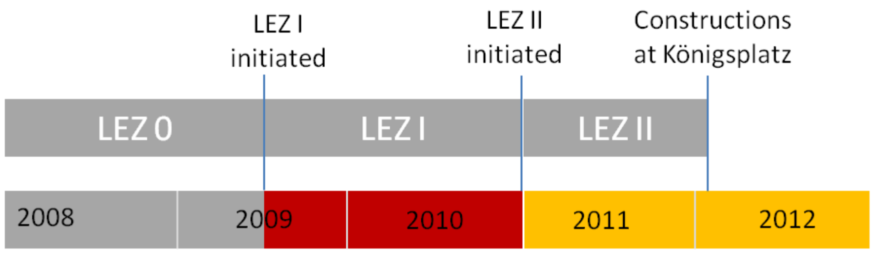



**Figure 13. Depiction of Low Emission Zone stages in Augsburg and time periods chosen for the analysis.**

The first stage of LEZ was introduced on July 1, 2009, and the second stage of LEZ became effective on January 1, 2011. The first stage of LEZ lasted for one and a half years (LEZ I). We chose the time period from January 1, 2008 to June 30, 2009 as the before LEZ period (LEZ 0), the same length as LEZ I. The LEZ II period was chosen between January 1, 2011 and February 28, 2012, because a massive construction work at Königsplatz (where one of the monitoring sites is located) in Augsburg started in March, 2012. To avoid the undesirable impact of the construction work on $PM_{10}$ concentrations, we analysed only data which were collected before the construction period.

#### 6.3.1.2 Descriptive analysis

Figure 14 shows the yearly mean concentrations of $PM_{10}$ from 2004 to 2012 in Augsburg. $PM_{10}$ concentrations were the highest at two traffic sites (KP and KS), followed by $PM_{10}$ at BP site, an urban background site in the inner city. The lowest $PM_{10}$ concentration was observed at LfU site, the background site located 4 km south of the city. Despite of the absolute differences in $PM_{10}$ levels, similar yearly trends were observed for all four monitoring sites: $PM_{10}$ concentrations were high in 2004 to 2006, from 2006 to 2007 pronounced decreases in $PM_{10}$ concentrations were observed and from 2008 to 2012 $PM_{10}$ concentrations decreased slightly in traffic sites, whereas an increase was observed in background sites. We compared $PM_{10}$ concentrations at the LfU site with another background site located in Munich, which is about 60 km southeast of Augsburg. The yearly mean concentrations of $PM_{10}$ at both background sites were shown in Figure 15. The $PM_{10}$ time series at both sites showed the same yearly trend, indicating that the yearly variation was due to the variation in regional background $PM_{10}$ concentration.

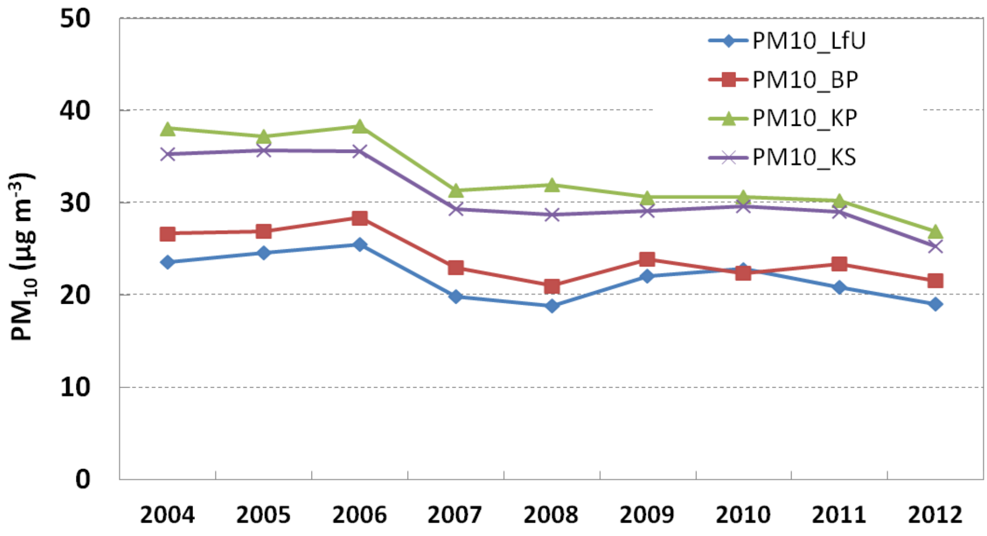



**Figure 14. Yearly mean concentrations of $PM_{10}$ at four monitoring sites in Augsburg from 2004 to 2012 (note: $PM_{10}$ concentrations at BP site were missing from January to June, 2010).**

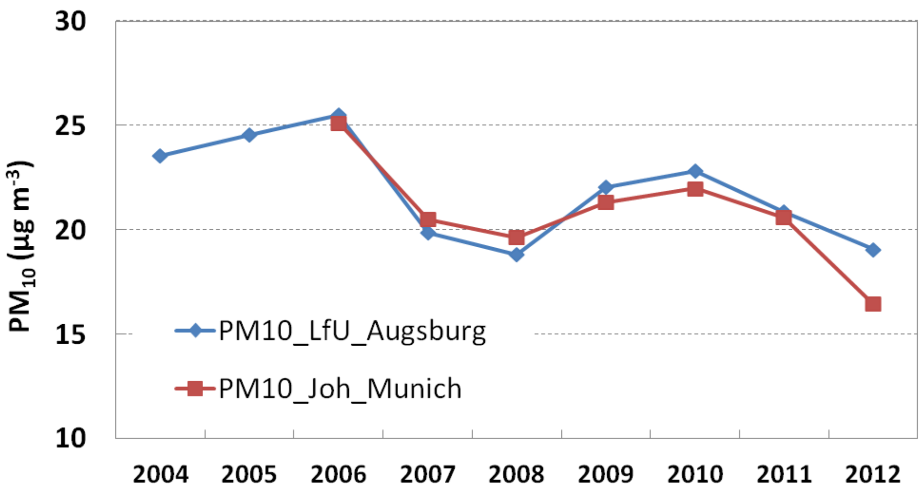



**Figure 15. Yearly mean concentrations of $PM_{10}$ at LfU site (Augsburg) and Johanneskirchen site (Munich) from 2004 to 2012.**

#### 6.3.1.3 Statistical analysis

The statistical model applied in Augsburg follows equation (2).

$$\begin{aligned}\log(PM10_{\mathrm{X}}) = {} & \beta_0 + \beta_1 \log(PM10_{\mathrm{ref}}) + \beta_{\mathrm{W0}} \cdot \mathrm{I}_{\mathrm{W0}} + \beta_{\mathrm{S1}} \cdot \mathrm{I}_{\mathrm{S1}} + \beta_{\mathrm{W1}} \cdot \mathrm{I}_{\mathrm{W1}} + \beta_{\mathrm{S2}} \cdot \mathrm{I}_{\mathrm{S2}} \\ & + \beta_{\mathrm{W2}} \cdot \mathrm{I}_{\mathrm{W2}} + f_{\mathrm{S0}} \cdot \mathrm{I}_{\mathrm{S0}}(hour) + f_{\mathrm{W0}} \cdot \mathrm{I}_{\mathrm{W0}}(hour) + f_{\mathrm{S1}} \cdot \mathrm{I}_{\mathrm{S1}}(hour) + \\ & f_{\mathrm{W1}} \cdot \mathrm{I}_{\mathrm{W1}}(hour) + f_{\mathrm{S2}} \cdot \mathrm{I}_{\mathrm{S2}}(hour) + f_{\mathrm{W2}} \cdot \mathrm{I}_{\mathrm{W2}}(hour) + \\ & f_{\mathrm{wd}}(wind\ direction) + \beta_2\, public\ holiday + \varepsilon.\end{aligned} \quad (2)$$

According to (2), the $PM_{10}$ concentrations at a specific monitoring site were calculated for “summer without LEZ, S0”, “summer LEZ I, S1”, “summer LEZ II, S2”, as well as “winter without LEZ, W0”, “winter LEZ I, W1”, and “winter LEZ II, W2", respectively. The relative changes of $PM_{10}$ in period 1 (LEZ I) and period 2 (LEZ II) compared with period 0 (without LEZ) in each season and for each site were calculated. As reference site the LfU site was chosen. The result will be presented in the following for each of the three monitoring sites in Augsburg.

#### 6.3.1.4 Königsplatz (KP)

Table 5 shows a summary of model result at KP site in Augsburg. The $PM_{10}$ concentration in summer without LEZ (S0) period was set as reference. $PM_{10}$ concentrations in other seasons and in other LEZ periods were modelled and effect estimates with respect to the reference period were shown. Figure 16 shows the effect estimates and the relative difference (as multiplicative factors) of $PM_{10}$ concentrations in different seasons and LEZ stages when comparing with $PM_{10}$ in summer without LEZ (S0) period.

In summer season, LEZ II had a reducing effect of 12% for $PM_{10}$, and LEZ I had a reducing effect of 11%. $PM_{10}$ in winter LEZ 0 was 11% higher than in summer LEZ 0. Decreasing effects in $PM_{10}$ concentration were observed for both winter LEZ I and II periods when compared the effect of winter LEZ 0 (111.9% vs. 99.0% and103.3%), however, the reduction effect turned out to be higher in winter LEZ I than winter LEZ II.

**Table 5. Summary of model results at KP site in Augsburg.**

| Parameter | Estimate | Multiplicative factor (%)* | Standard Error | p Value |
|---|---|---|---|---|
| Intercept | 2.307 | | 0.0232 | <0.001 |
| Log(**$PM_{10}$**_LfU) | 0.343 | | 0.0048 | <0.001 |
| Public Holiday | -0.082 | 92.1% | 0.0325 | 0.011 |
| Summer, LEZ 0 | reference | | | |
| Summer, LEZ I | -0.118 | 88.9% | 0.0277 | <0.001 |
| Summer, LEZ II | -0.132 | 87.6% | 0.0281 | <0.001 |
| Winter, LEZ 0 | 0.104 | 111.0% | 0.0278 | <0.001 |
| Winter, LEZ I | -0.010 | 99.0% | 0.0277 | 0.730 |
| Winter, LEZ II | 0.032 | 103.3% | 0.0278 | 0.246 |

*The multiplicative factors are the ratios of $PM_{10}$ mass concentrations in different LEZ stages and seasons compared to $PM_{10}$ in summer during LEZ 0.

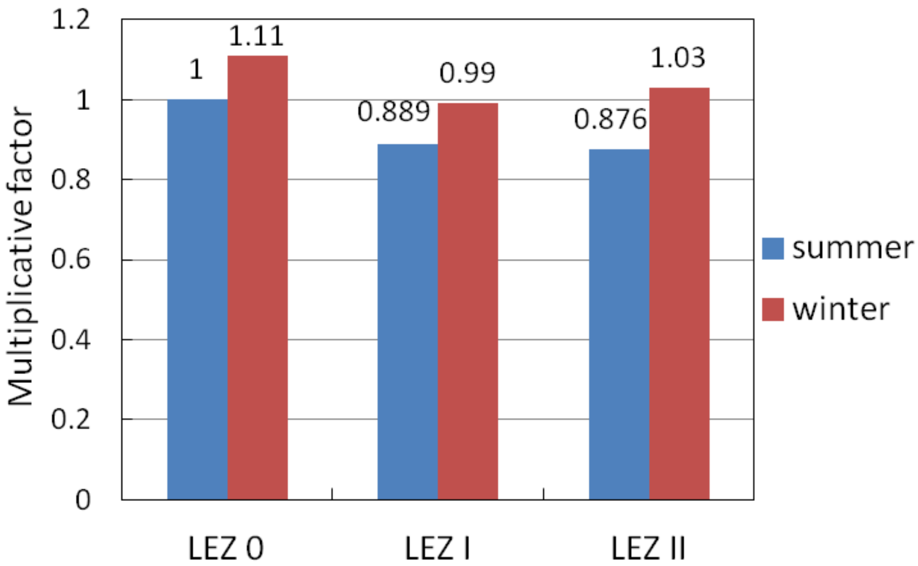



**Figure 16. Change of $PM_{10}$ concentrations in LEZ I and LEZ II when compared to LEZ 0 at KP site in Augsburg (the reference is summer LEZ 0).**

#### 6.3.1.5 Karlstrasse (KS)

Table 6 gives a summary of model result at KS site in Augsburg. As in the model for KP site, the $PM_{10}$ concentration in summer without LEZ (S0) was set as reference. $PM_{10}$ concentrations in other seasons and for other LEZ stages were modelled and factors with respect to the reference $PM_{10}$ were shown. Figure 17 illustrates the factors of $PM_{10}$ concentrations in different seasons and LEZ stages when comparing with $PM_{10}$ in summer LEZ 0 period.

In summer season, $PM_{10}$ concentrations in LEZ II and LEZ I were 92% and 93%, respectively of the concentrations in LEZ 0 period. This indicates effectiveness of LEZ in summer season in reducing the $PM_{10}$ concentration, though the difference is very small between summers LEZ I and LEZ II. $PM_{10}$ in winter LEZ 0 was 6% higher than in summer LEZ 0. Increasing effects were observed for winter LEZ I and II periods (106.5% vs. 109.7% and 116.3%).

**Table 6. Summary of model results at KS site in Augsburg.**

| Parameter | Estimate | Multiplicative factor (%) * | Standard Error | P Value |
|---|---|---|---|---|
| Intercept | 2.459 | | 0.0270 | <0.001 |
| Log(**$PM_{10}$**_LfU) | 0.235 | | 0.0057 | <0.001 |
| Public Holiday | -0.191 | 82.6% | 0.0384 | <0.001 |
| Summer, LEZ 0 | reference | | | |
| Summer, LEZ I | -0.069 | 93.3% | 0.0315 | 0.029 |
| Summer, LEZ II | -0.078 | 92.5% | 0.0325 | 0.017 |
| Winter, LEZ 0 | 0.063 | 106.5% | 0.0321 | 0.049 |
| Winter, LEZ I | 0.093 | 109.7% | 0.0320 | 0.0036 |
| Winter, LEZ II | 0.151 | 116.3% | 0.0321 | <0.001 |

**The multiplicative factors are the ratios of* $PM_{10}$ *mass concentrations in different LEZ stages and seasons compared to* $PM_{10}$ *in summer during LEZ 0.*

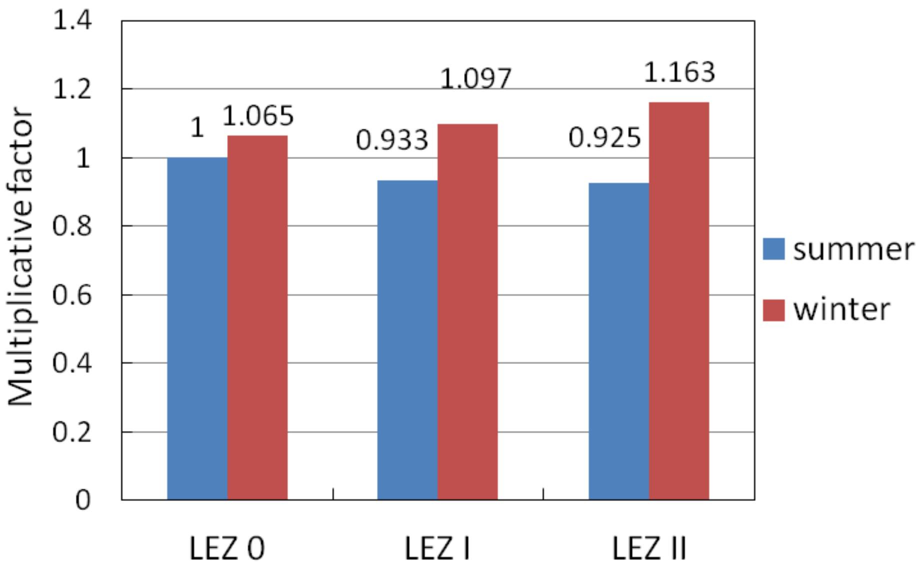



**Figure 17. Change of $PM_{10}$ concentration in LEZ I and LEZ II when compared to LEZ 0 at KS site (the reference is summer LEZ 0).**

#### 6.3.1.6 Bourgesplatz (BP)

A summary of model for BP is presented in Table 7. Figure 18 illustrates the factors of $PM_{10}$ concentrations in different seasons and LEZ stages when comparing with $PM_{10}$ in summer LEZ 0 period.

The p-values for summer seasons were larger than 0.5, indicating that $PM_{10}$ in summer LEZ I and LEZ II were not significantly changed compared with summer LEZ 0, thus no effectiveness was observed for LEZ in summer season. In winter LEZ 0, $PM_{10}$ was 12% higher than in summer LEZ 0. $PM_{10}$ in winter LEZ I and LEZ II were higher than winter LEZ 0 (118.2% in winter LEZ I, 132.9% in winter LEZ II).

**Table 7. Summary of model results at Bourgesplatz site in Augsburg.**

| Parameter | Estimate | Multiplicative factor (%)* | Standard Error | P Value |
|---|---|---|---|---|
| Intercept | 2.199 | | 0.040 | <0.001 |
| Log($PM_{10}$_LfU) | 0.198 | | 0.005 | <0.001 |
| Public Holiday | -0.089 | 91.5% | 0.039 | 0.0214 |
| Summer, LEZ 0 | | | | |
| Summer, LEZ I | -0.012 | 98.8% | 0.062 | 0.843 |
| Summer, LEZ II | 0.020 | 102.0% | 0.052 | 0.699 |
| Winter, LEZ 0 | 0.113 | 112.0% | 0.053 | 0.0336 |
| Winter, LEZ I | 0.167 | 118.2% | 0.059 | 0.0048 |
| Winter, LEZ II | 0.285 | 132.9% | 0.053 | <0.001 |

**The multiplicative factors are the ratios of $PM_{10}$ mass concentrations in different LEZ stages and seasons compared to $PM_{10}$ in summer during LEZ 0.*

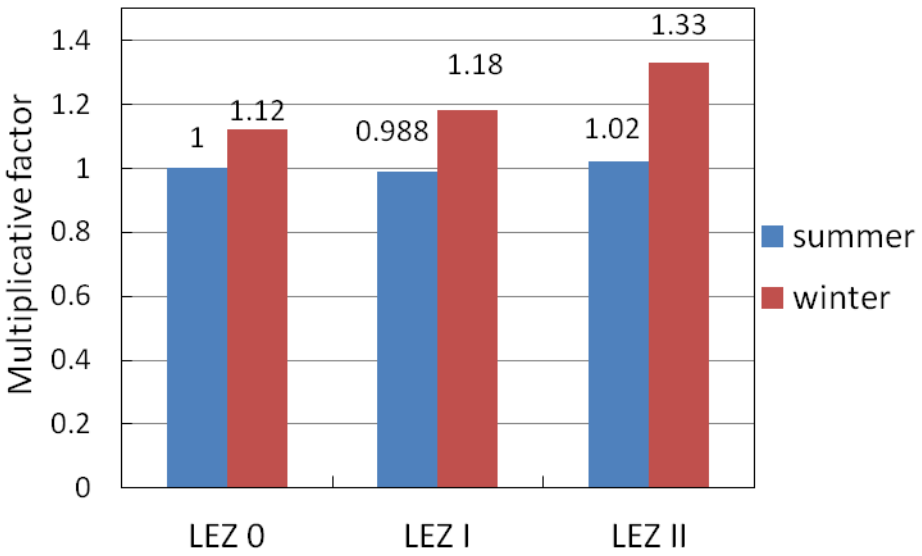



**Figure 18. Change of $PM_{10}$ concentrations in LEZ I and LEZ II when compared to LEZ 0 at BP site (the reference is LEZ 0 in summer).**

#### 6.3.1.7 Summary of the results of statistical analysis for Augsburg

A summary of the changes on $PM_{10}$ in Augsburg is shown in **Fel! Hittar inte referenskälla.**. In Augsburg the LEZ effects were inconsistent between summer and winter, as well as between traffic sites and urban background site. In the summer season a decrease of $PM_{10}$ levels was observed for the two traffic sites (KP and KS), whereas no significant effect of LEZ was observed for the urban background site BP. In the winter season, we observed decrease of $PM_{10}$ concentrations only for KP site, while even an increase of $PM_{10}$ levels was observed at the KS and BP sites. It seems that the LEZ has an impact on the $PM_{10}$ concentrations in Augsburg, but only in the summer season and only at traffic sites. No effects of LEZ were detected in the summer season at the urban background site BP. Due to the inconsistent results we conclude that the impact of the LEZ in Augsburg on the $PM_{10}$ levels is rather weak, if any.

**Table 8. Change of $PM_{10}$ concentration[a] in Augsburg in period 1 and period 2 when compared with period 0 (without any measures) at Königsplatz, Karlsstrasse, and Bourgesplatz.**

| Measurement site | Summer | | | Winter | | |
|---|---|---|---|---|---|---|
| Period 1 (LEZ I) | | | | | | |
| | Effect | Confidence interval | p-value | Effect | Confidence interval | p-value |
| Königsplatz | **-11.1%** | (-20.0%, -1.4%) | <0.001 | **-10.8%** | (-19.6%, -0.9%) | <0.001 |
| Karlstrasse | **-6.6%** | (-12.2%, -0.7%) | 0.03 | 3.0% | (-3.2%, 9.7%) | <0.05 |
| Bourgesplatz | -1.2% | (-12.6%, 11.6%) | 0.84 | 5.5% | (-6.0%, 18.5%) | <0.005 |
| Period 2 (LEZ II) | | | | | | |
| Königsplatz | **-12.4%** | (-21.1%, -2.7%) | <0.001 | **-7.0%** | (-16.2%, 3.3%) | <0.001 |
| Karlstrasse | **-7.5%** | (-13.2%, -1.4%) | 0.02 | 9.2% | (2.5%, 16.2%) | <0.001 |
| Bourgesplatz | 2.0% | (-7.9%, 13.1%) | 0.70 | 18.8% | (7.0%, 31.8%) | <0.001 |

[a] *adjusted for* $PM_{10}$ *concentration at the reference station, wind direction, day of the week, time of the day and public holidays.*

### 6.3.2 LEZ and transit ban effects on $PM_{10}$ in Munich

#### 6.3.2.1 Measurement sites and study period in Munich

Figure 19 shows the location of all LÜB monitoring sites operated at the time of the implementation of the LEZ in Munich. However, the monitoring sites at Prinzregentenstrasse and Luise-Kieselbach-Platz were closed some time ago and the date collected at those sites could not be used for the analysis. We used the $PM_{10}$ data collected at three monitoring sites. Those sites include one urban background site, Lothstrasse (measurement height: 4 m over ground) (Regierung von Oberbayern, 2007; 2010) one street site on the border of LEZ, Landshuter Allee (131 000 vehicles/day), and one street site in the city centre, Stachus. The local traffic contributes 6% to $PM_{10}$ the site Lothstrasse, and 44% to $PM_{10}$ at site Landshuter Allee (Regierung von Oberbayern, 2010). A regional background site in the outskirts of the city and outside the LEZ in Johanneskirchen was selected as a reference site (measurement height: 4 m over ground; distance to road: 5 m) (Regierung von Oberbayern, 2010).

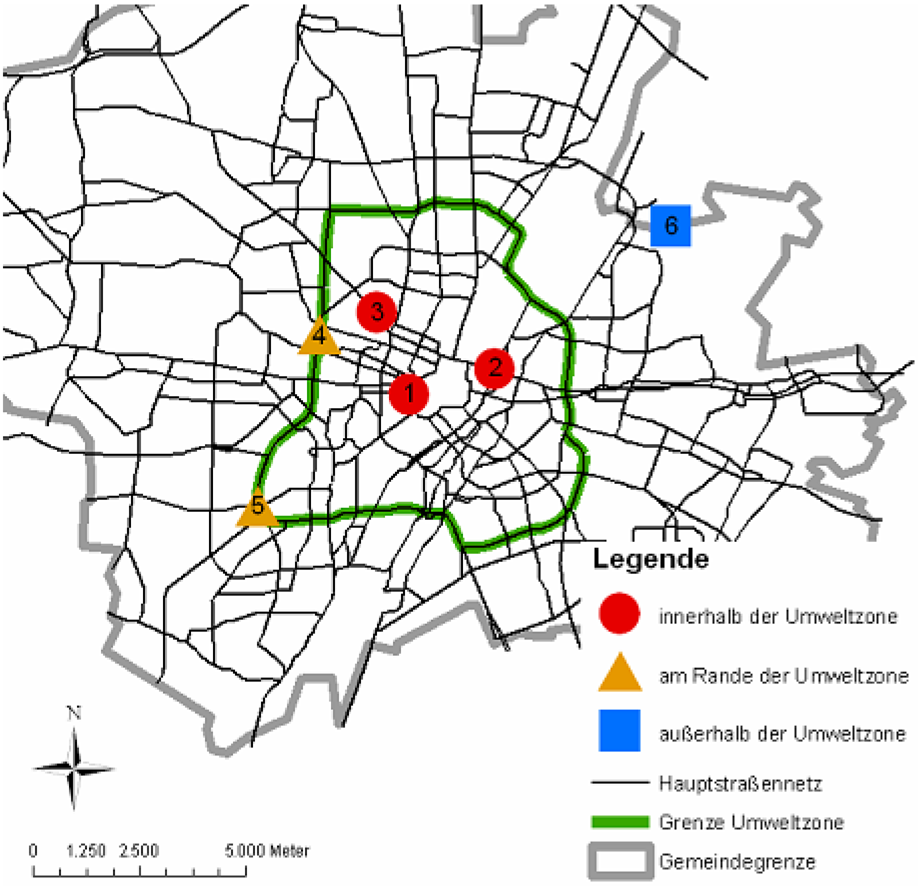



**Figure 19. Locations of the LÜB monitoring sites in Munich: Stachus (1), Prinzregentenstrasse (2), Lothstrasse (3), Landshuter Allee (4), Luise-Kieselbach-Platz (5), Johanneskirchen (6).**

We set up three time periods depending on different traffic control measures. Before February 1, 2008, there was no traffic control measures introduced in Munich, and the time period between February 1, 2006 and January 31, 2008 was defined as period 0 (without measures). In the period between February 1, 2008 and September 30, 2008, only a truck transit ban was effective. This period was excluded from the analysis because it was too short to obtain statistically significant results. The examination period 1 (both transit ban for heavy-duty vehicles and LEZ in stage 1 were effective) covered the time between October 1, 2008 and September 30, 2010. The period 2 was chosen between October 1, 2010 and September 30, 2012 (transit ban and LEZ II). From October 1, 2010, the second stage of LEZ became effective. Vehicles with emission standards of Euro 2 or less were not allowed to enter the LEZ. The transit ban for heavy-duty vehicles was still in operation in period 2. The $PM_{10}$ values on January 1 were always excluded from the analysis due to the impact from fireworks on New Year's Eve.

#### 6.3.2.2 Descriptive analysis

Figure 20 shows the yearly mean of $PM_{10}$ concentrations at four LÜB monitoring sites in Munich. $PM_{10}$ concentrations decreased from 2006 to 2008, and remained at the same level, or increased slightly from 2008 to 2010. It decreased again from

2010 till 2012. $PM_{10}$ measured at all four sites showed the same yearly trends, though the concentration levels differed, due to the different site types (Johanneskirchen: regional background, Lothstrasse: urban background, Stachus and Landshuter Allee: traffic sites).

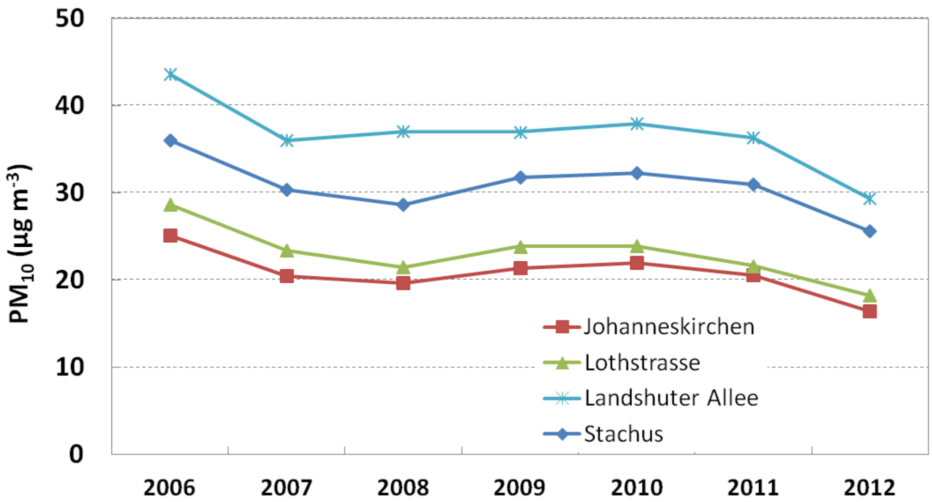



**Figure 20. Yearly mean $PM_{10}$ concentrations at four monitoring sites in Munich from 2006 to 2012.**

#### 6.3.2.3 Statistical analysis

The statistical model applied for Munich follows equation (2) – the same as for Augsburg.

$PM_{10}$ concentrations at a specific monitoring site were calculated for "summer without measures", "summer LEZ I & transit ban", "summer LEZ II & transit ban", as well as "winter without measures", "winter LEZ I & transit ban", and "winter LEZ II & transit ban", respectively. The relative changes of $PM_{10}$ in period 1 (LEZ I & transit ban) and period 2 (LEZ II & transit ban) compared with period 0 (without measures) in each season and for each site were calculated. The reference site is the Johanneskirchen site. The results are presented in the following for each of the three monitoring sites in Munich separately.

#### 6.3.2.4 Landshuter Allee

Figure 21 shows the temporal patterns of the modelled $PM_{10}$ concentrations at Landshuter Allee for the periods with and without measures (adjusted for $PM_{10}$ concentration at the reference station, wind direction and public holidays). $PM_{10}$ in summer showed a diurnal variation with major peak in the morning and a second peak in the evening. In winter $PM_{10}$ concentrations were high during the daytime. The diurnal pattern was less pronounced on weekends in both summer and winter seasons.

The efficiency of the measures depended on the time of the day and followed a diurnal pattern. The $PM_{10}$ burden was reduced more efficiently during hours with higher relative and absolute $PM_{10}$ mass concentration, i.e. between the morning and afternoon rush hours.

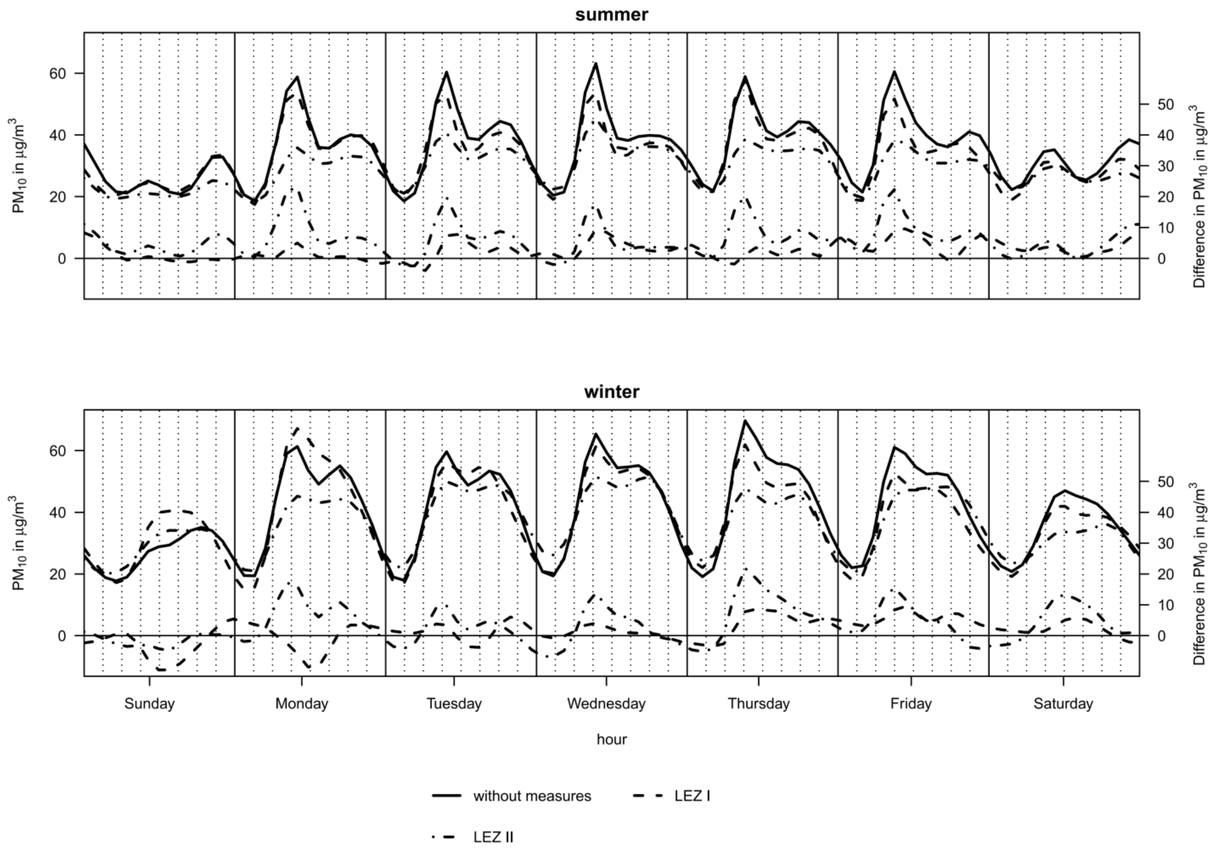



**Figure 21. Modeled hourly concentrations of $PM_{10}$ at Landshuter Allee adjusted for $PM_{10}$ at the reference station, wind direction and public holidays. Differences in $PM_{10}$ concentrations are shown on the right axis (positive signifies decrease by LEZ implementation).**

Figure 22 shows the daily mean effects of both measures (transit ban and LEZ) in period 1 and period 2, respectively at Landshuter Allee (differentiated by season and weekday). Period 2 had stronger effect in weekdays than period 1 in both summer and winter. The effect of the measures for each weekday was stronger in the summer season than in winter, especially in period 2. Stronger effects were observed on Fridays and Saturdays. On Sundays a strong seasonal dependency of the effect was observed: the measures were only effective in summer.

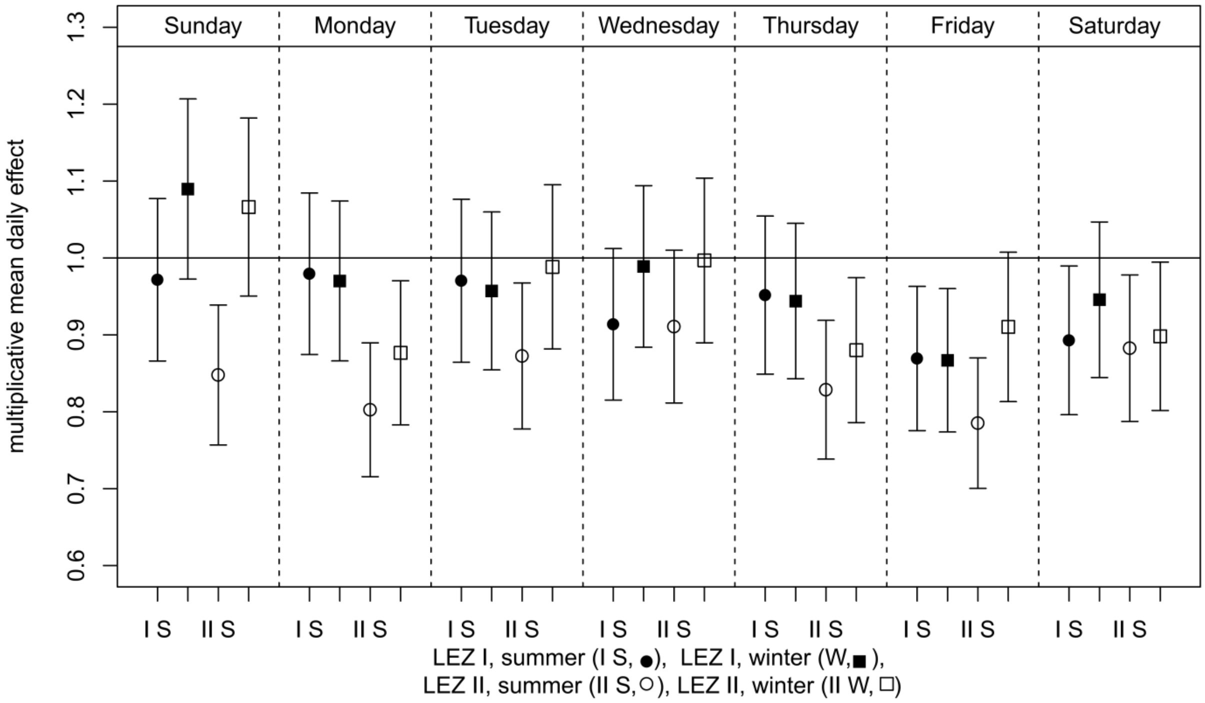



**Figure 22. Daily mean effects of the measures at Landshuter Allee stratified by season and day of the week with 95 % confidence intervals.**

#### 6.3.2.5 Stachus

Figure 23 shows the temporal patterns of the modelled $PM_{10}$ concentrations at Stachus for the periods 0, 1 and 2, as well as the differences in $PM_{10}$ between period 1 and 0, as well as period 2 and period 0. $PM_{10}$ showed a morning peak and a weaker evening peak in winter weekdays, while $PM_{10}$ concentrations were high during the daytime in winter season. The $PM_{10}$ daily peaks were much lower than in Landshuter Allee, indicating that Stachus was less affected by rush-hour traffic. The traffic intensity at Stachus is lower and more constant over the whole day compared with Landshuter Allee site.

At Stachus, the effect of the traffic control measures in period 1 was not detected by the model. $PM_{10}$ concentrations were higher in period 1 than period 0 from Monday to Thursday in summer, and from Monday to Wednesday in winter. Note that at the station Stachus, a construction and tram track maintenance was conducted between August 2008 and April 2010 in the vicinity of this monitoring site. Thus in period 1, $PM_{10}$ concentration at Stachus was probably influenced (increased) by the construction work. $PM_{10}$ concentrations in period 2 were lower than in period 0 for most of the time, but the reduction was small. Overall, the efficiency of the traffic control measures was lower at Stachus than at Landshuter Allee.

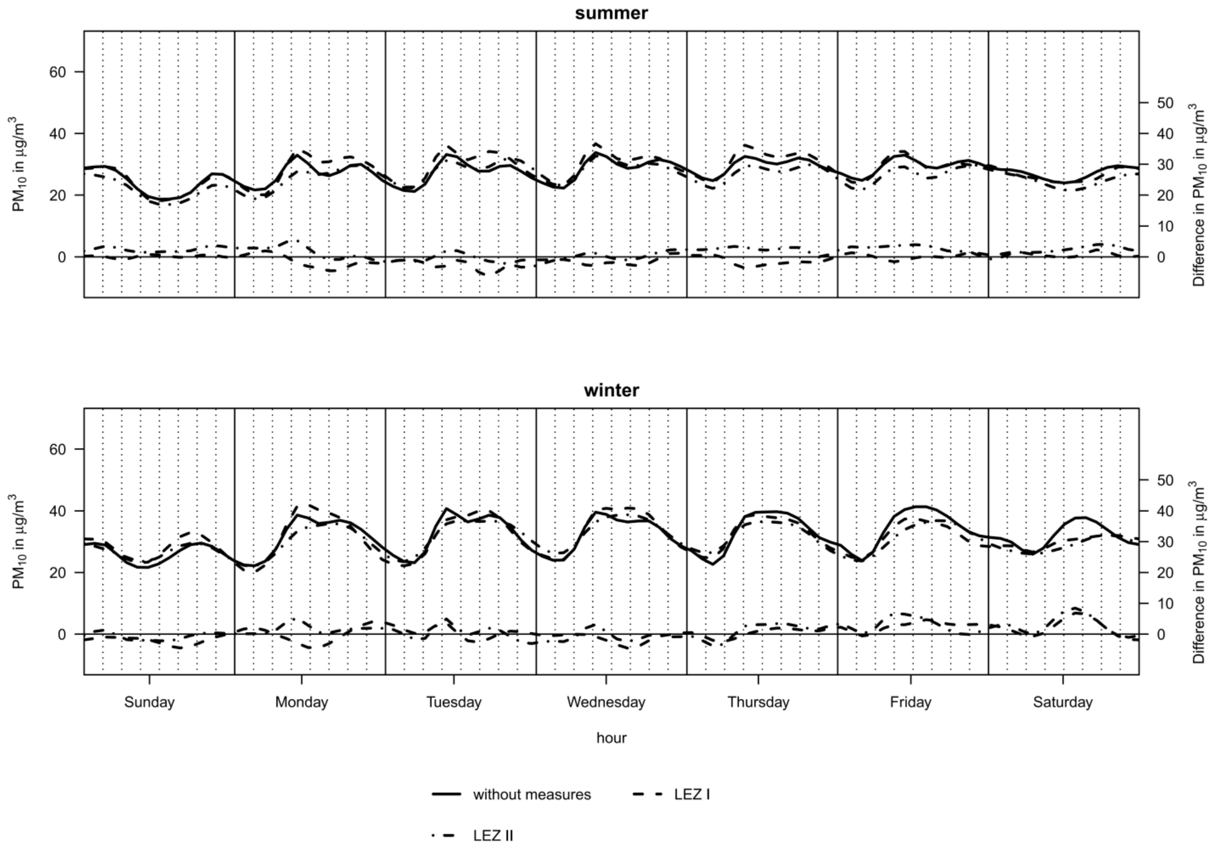



**Figure 23. Modeled hourly concentrations of $PM_{10}$ at Stachus adjusted for $PM_{10}$ at the reference station, wind direction and public holidays. Differences in $PM_{10}$ concentrations are shown on the right axis (positive signifies decrease by LEZ implementation).**

Figure 24 gives the daily mean effects of both measures (transit ban and LEZ) at Stachus in period 1 and period 2, respectively. The efficiency of control measures in period 1 was not seen from Sunday to Thursday, due to the increase in $PM_{10}$ concentration from construction. On Friday and Saturday in period 1, a decrease in $PM_{10}$ was observed despite of construction. In period 2, decrease of $PM_{10}$ was observed in most of the weekdays except Tuesday and Wednesday, and winter Sunday. When focusing on period 2, we found significant effects of transit ban and LEZ II control measures in reducing $PM_{10}$ concentration by 5.6% in summer and 5.7% in winter at Stachus.

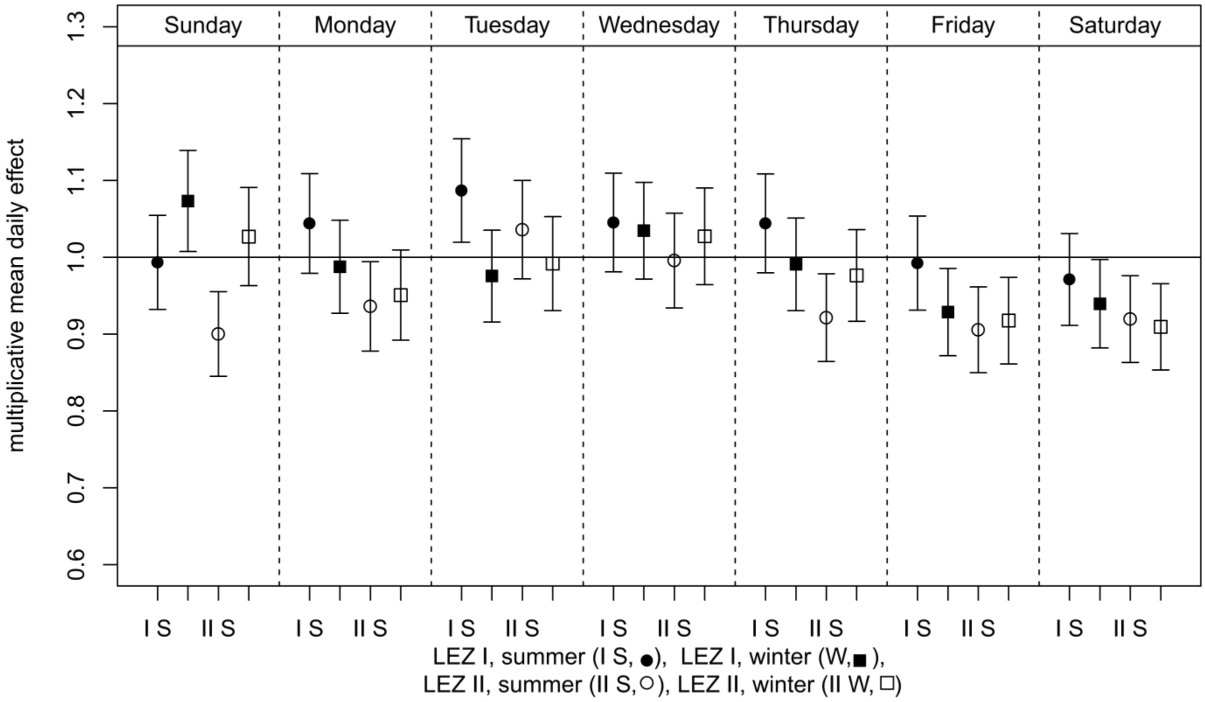



**Figure 24. Daily mean effects of the measures at Stachus stratified by season and day of the week with 95 % confidence intervals.**

#### 6.3.2.6 Lothstrasse

The temporal patterns of the modelled $PM_{10}$ concentrations at Lothstrasse for the periods with and without measures are shown in Figure 25. Figure 26 shows the daily mean effects of both measures (transit ban and LEZ) at Lothstrasse in period 1 and period 2, respectively. There is almost not diurnal variation of $PM_{10}$ at Lothstrasse both in summer and winter season.

Overall, $PM_{10}$ concentrations at Lothstrasse were significantly reduced in periods with control measures. $PM_{10}$ concentration was reduced by 6.4% (summer) and 5.0% (winter) in period 1 and 10.0% (summer) and 6.4% (winter) in period 2 compared to period 0 (reference period). Higher reduction in $PM_{10}$ was observed on Monday, Friday and Saturday. The effects on Sunday were dependent on season: measures were only effective in summer.

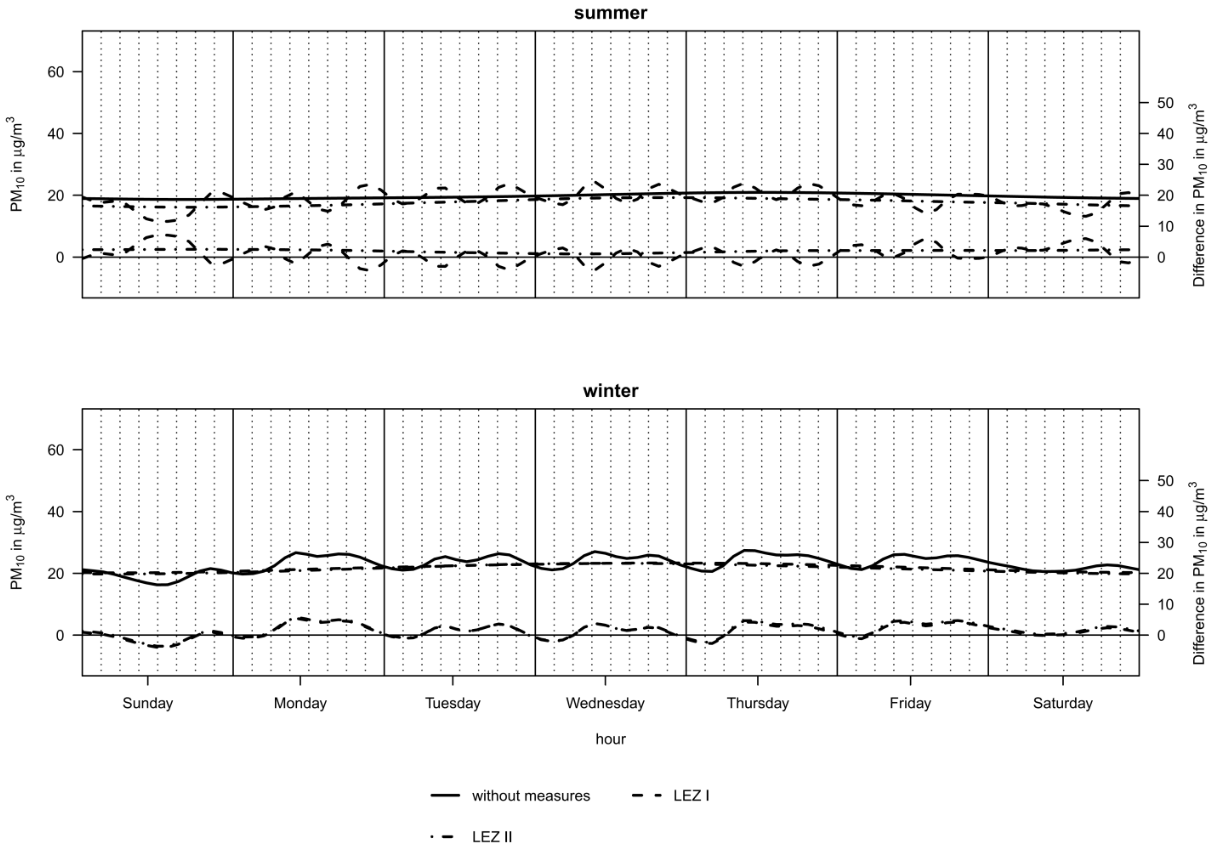



**Figure 25. Modeled hourly concentrations of $PM_{10}$ at Lothstrasse adjusted for $PM_{10}$ at the reference station, wind direction and public holidays. Differences in $PM_{10}$**

**concentrations are shown on the right axis (positive signifies decrease by LEZ implementation).**

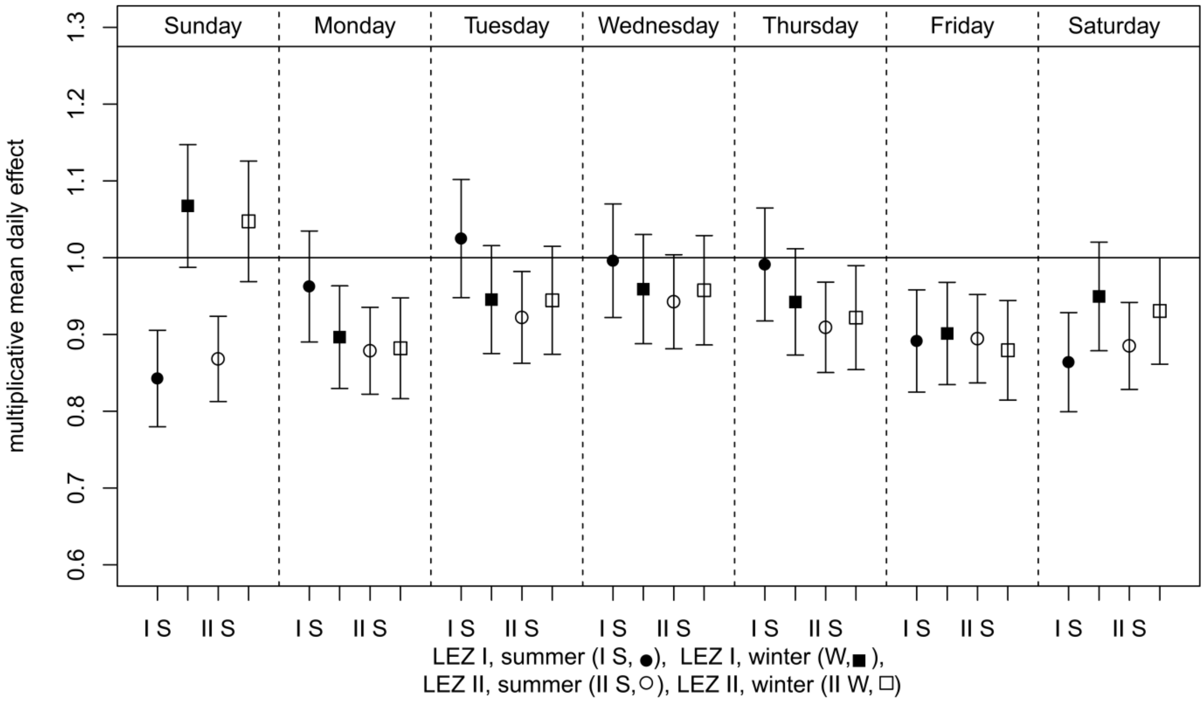



**Figure 26. Daily mean effects of the measures stratified by season and day of the week with 95 % confidence intervals for Lothstrasse.**

#### 6.3.2.7 Summary of the results of statistical analysis for Munich

A summary of the changes on $PM_{10}$ in Munich is shown in Table 9. Note that the effectiveness was a combination of both LEZ and transit ban for heavy duty vehicles. LEZ and transit ban control measures significantly reduced the $PM_{10}$ concentrations in Munich.

**Table 9. Change of $PM_{10}$ concentration[a] in Munich in period 1 and period 2 when compared with period 0 (without any measures) at Lothstrasse, Landshuter Allee, and Stachus.**

| Measurement site | Summer | | | Winter | | |
|---|---|---|---|---|---|---|
| Period 1 (LEZ I and transit ban) | | | | | | |
| | Effect | Confidence interval | p-value | Effect | Confidence interval | p-value |
| Lothstr. | **-6.4%** | (-9.6%, -3.1%) | <0.001 | **-5.0%** | (-8.3%, -1.7%) | 0.004 |
| Landshuter Alle | **-6.5%** | (-10.9%, -2.2%) | 0.004 | -3.6% | (-8.0%, 0.8%) | 0.1150 |
| Stachus | 2.5% | (-0.1%, 5.0%) | 0.054 | -1.1% | (-3.5%, 1.3%) | 0.372 |
| Period 2 (LEZ II and transit ban) | | | | | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| Lothstr. | **-10.3%** | (-13.2%, -6.9%) | <0.001 | **-6.4%** | (-9.7%, -3.1%) | <0.001 |
| Landshuter Alle | **-15.4%** | (-19.3%, -11.5%) | <0.001 | **-5.7%** | (-10.1%, -1.4%) | 0.0120 |
| Stachus | **-5.6%** | (-8.0%, -3.3%) | <0.001 | **-5.7%** | (-5.4%, -0.6%) | 0.018 |

[a] *adjusted for* $PM_{10}$ *concentration at the reference station, wind direction, day of the week, time of the day and public holidays.*

For all sites, decreases in $PM_{10}$ concentrations were observed except Stachus site in period 1. Summer seasons generally had higher efficiency than in winter seasons (except Stachus). Stronger effects were found in period 2 than in period 1. The decrease in $PM_{10}$ ranged from 3.6% to 6.5% in period 1, and 5.7%-15.4% in period 2. The highest efficiency in reducing $PM_{10}$ concentration was in summer in period 2 at Landshuter Allee. In contrast, at Stachus site in period 1, due to the impact of nearby construction activity, no effect was observed.

### 6.3.3 LEZ effects on $PM_{10}$ in Berlin

#### 6.3.3.1 Measurement sites and study period in Berlin

$PM_{10}$ concentrations measured at three monitoring stations were used in statistical model (Figure 27). The monitoring sites belong to the Berlin Air Quality Monitoring Network (Berliner Luftgüte-Messnetzes, BLUME, http://www.berlin.de/umwelt/aufgaben/luft-messnetz.html). Frankfurter Allee is a traffic site, located in a densely populated residential area in the inner city and on the road of Frankfurter Allee, a road with high traffic density. Schildhornstr site is a traffic site, located in a street with high traffic volume in a residential area in the inner city. Nansenstr site is an urban background site, located in a residential area in the inner city, within the LEZ. Due to little impact from local traffic, Nansenstr site is selected as the reference site.

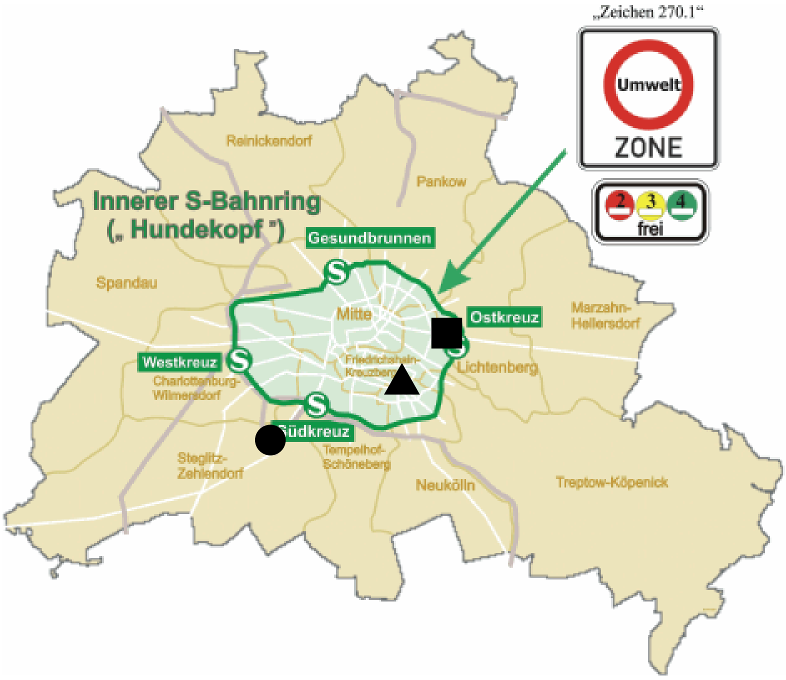



**Figure 27. Locations of monitoring sites in Berlin (Schildhornstr (●), Frankfurter Allee (■), Nansenstr (▲)) (http://www.berlin.de/umwelt/aufgaben/verkehr-umweltzone.html).**

Three time periods were separated from 2004 till 2012 in Berlin. Period 1 is from January 1, 2008 to December 31, 2009, corresponding to the first stage of LEZ. Period 2 is from January 1, 2010 to December 31, 2012, when the third stage of LEZ was in effective. The time from January 1, 2004 to December 31, 2007 was defined as period 0, when no LEZ measure was introduced.

#### 6.3.3.2 Descriptive analysis

As shown in Figure 28, $PM_{10}$ concentrations had the same temporal variation among three stations. On average, $PM_{10}$ concentration was the lowest at Nansenstr, the background site, and the highest concentration was observed at Frankfurter Allee, the traffic site.

Table 10 gives the mean values of $PM_{10}$ concentrations between three periods. For all three monitoring sites, highest concentrations were in "No LEZ" period, and the lowest values in "LEZ stage III". This indicates the same trend in $PM_{10}$ concentrations at different types of sites in long term.

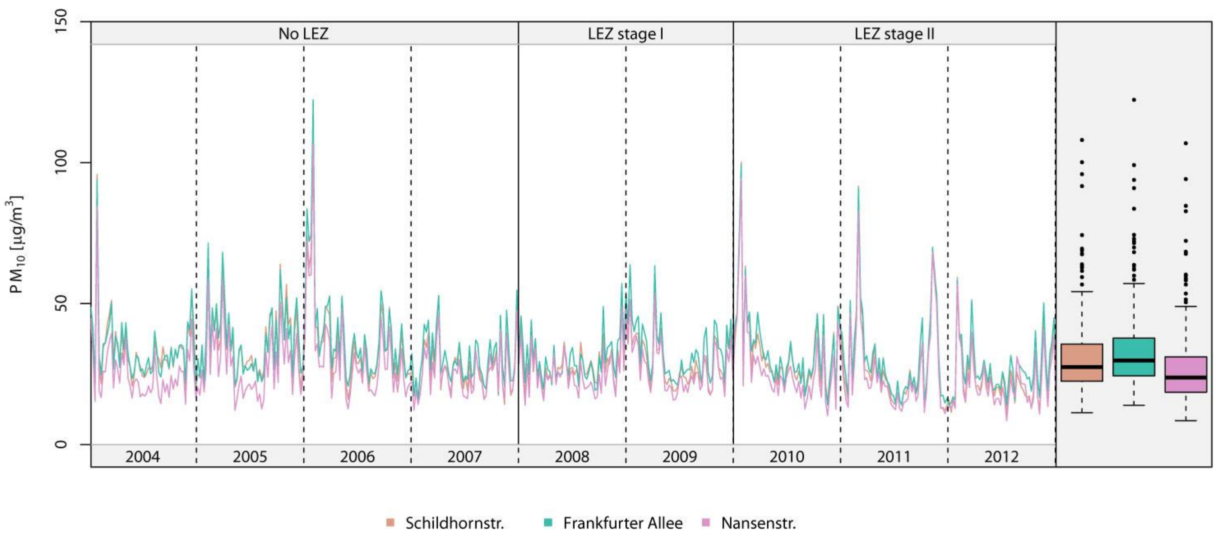



**Figure 28. Temporal variation of weekly $PM_{10}$ concentrations at three monitoring sites in Berlin from 2004 to 2012.**

**Table 10. Mean and standard deviation of $PM_{10}$ concentrations at three monitoring stations in Berlin in three LEZ stages.**

| | **No LEZ** | | **LEZ stage I** | | **LEZ stage III** | |
|---|---|---|---|---|---|---|
| Monitoring site | n | $PM_{10}$ mean(SD) | **n** | $PM_{10}$ mean(SD) | n | $PM_{10}$ mean(SD) |
| Schildhornstr. | 34060 | 33.1 (20.5) | 17252 | 28.5 (14.6) | 25903 | 28.0 (20.0) |
| Frankfurter Allee | 34369 | 35.1 (21.9) | 17266 | 31.9 (17.5) | 25874 | 30.3 (20.6) |
| Nansenstr. | 34557 | 27.5 (18.8) | 17248 | 26.1 (14.2) | 25764 | 25.0 (19.0) |

Note: the mean values were calculated from hourly $PM_{10}$ concentrations

#### 6.3.3.3 Statistical analysis

The statistical model used for Berlin is the same as the one used for Augsburg and Munich – see equation (2).

$PM_{10}$ concentrations at a specific monitoring site were calculated for "summer without LEZ", "summer LEZ I", "summer LEZ III", as well as "winter without LEZ", "winter LEZ I", and "winter LEZ III", respectively. The relative changes of $PM_{10}$ in period 1 (LEZ I) and period 2 (LEZ III) compared with period 0 (without LEZ) in each season and for each site were calculated. The reference site is the Nansenstrasse site. The result will be presented in the following for each of two monitoring sites in Berlin.

#### 6.3.3.4 Schildhornstr

Modelled $PM_{10}$ concentrations at Schildhornstr showed a significant diurnal variation on weekdays with a morning peak between 9:00 and 10:00 am. On weekends no morning peak was observed. $PM_{10}$ concentrations were lower in period 1 and 2 compared with period 0. In summer season, reduction of $PM_{10}$ was observed throughout the day, however, smaller reduction was around midnight until early morning, when $PM_{10}$ concentration was the lowest and the on-road traffic was the least within a day. In winter, reductions of $PM_{10}$ didn't show a diurnal trend (Figure 29).

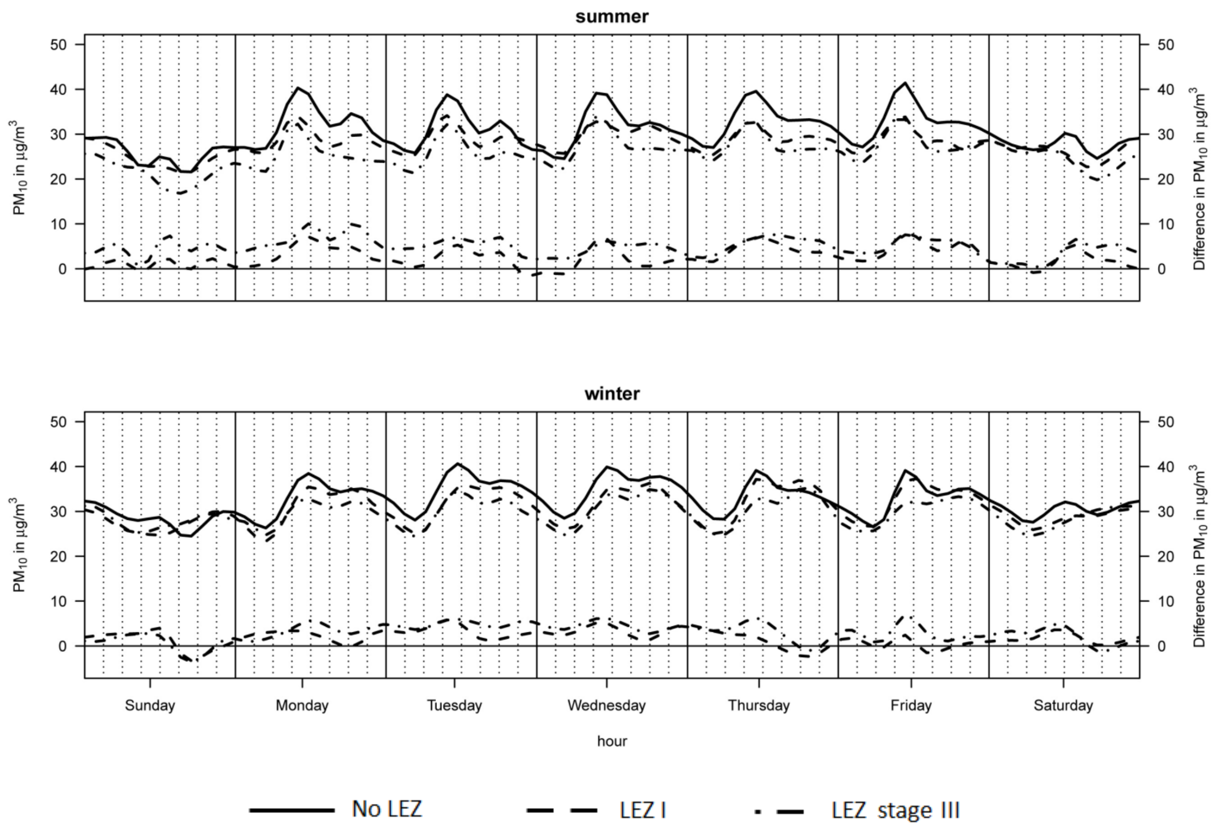



**Figure 29. Modelled hourly concentrations of $PM_{10}$ at Schildhornstr adjusted for $PM_{10}$ at the reference station, wind direction and public holidays. Differences in $PM_{10}$ concentrations are shown on the right axis (positive signifies decrease by LEZ implementation).**

Figure 30 shows the daily mean effects of LEZ I and LEZ III on $PM_{10}$ concentration at Schildhornstr. In LEZ stage I, the effects in summer was lower than in winter. In LEZ stage III, the effects were higher than in LEZ I. For each weekday, the effects differed from summer to winter and from LEZ I to III. Despite of this, effects were observed for all 7 days in one week.

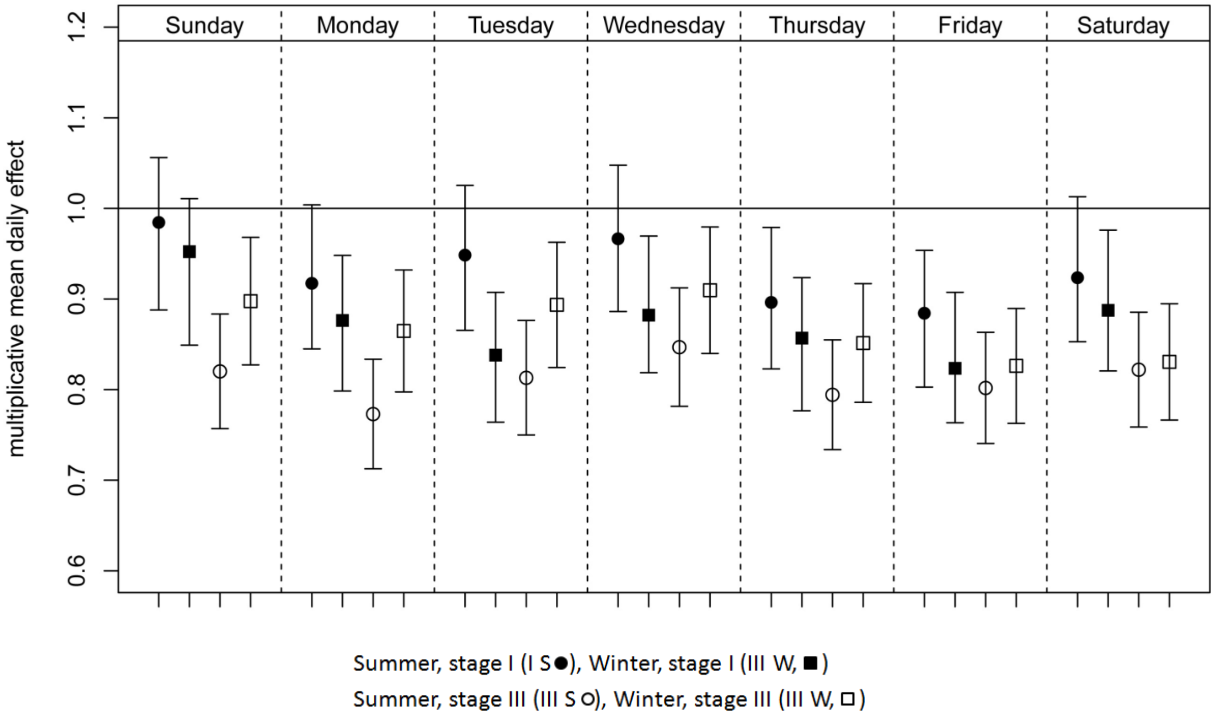



**Figure 30. Daily mean effects of the measures stratified by season and day of the week with 95 % confidence intervals for Schildhornstr.**

### 6.3.3.5 Frankfurter Allee

Figure 31 shows the modelled $PM_{10}$ concentrations at Frankfurter Allee in LEZ stage I and III, compared with no LEZ period. Consistent $PM_{10}$ diurnal variation was observed during weekdays. In summer, a morning peak (9:00 – 12:00 am) was observed, while in winter it showed a two-peak diurnal pattern, with on in the morning (9:00 – 12:00 am) and the other in the evening (6:00 – 9:00 pm). During weekends the diurnal variation was much weaker than on weekdays. The reductions in $PM_{10}$ in LEZ I and III compared with no LEZ period showed a diurnal variation in summer, with the strongest reduction in the morning, and the weakest reduction in the early morning. In winter, there is no diurnal pattern in the reduction of $PM_{10}$: effects were similar throughout the day.

As shown in Figure 32, effects of LEZ I and LEZ III in reducing $PM_{10}$ concentration were observed for all days in a week at Frankfurter Allee. Weaker effects were on Saturday and Sunday in summer season. The strongest effects were observed in summer in LEZ III.

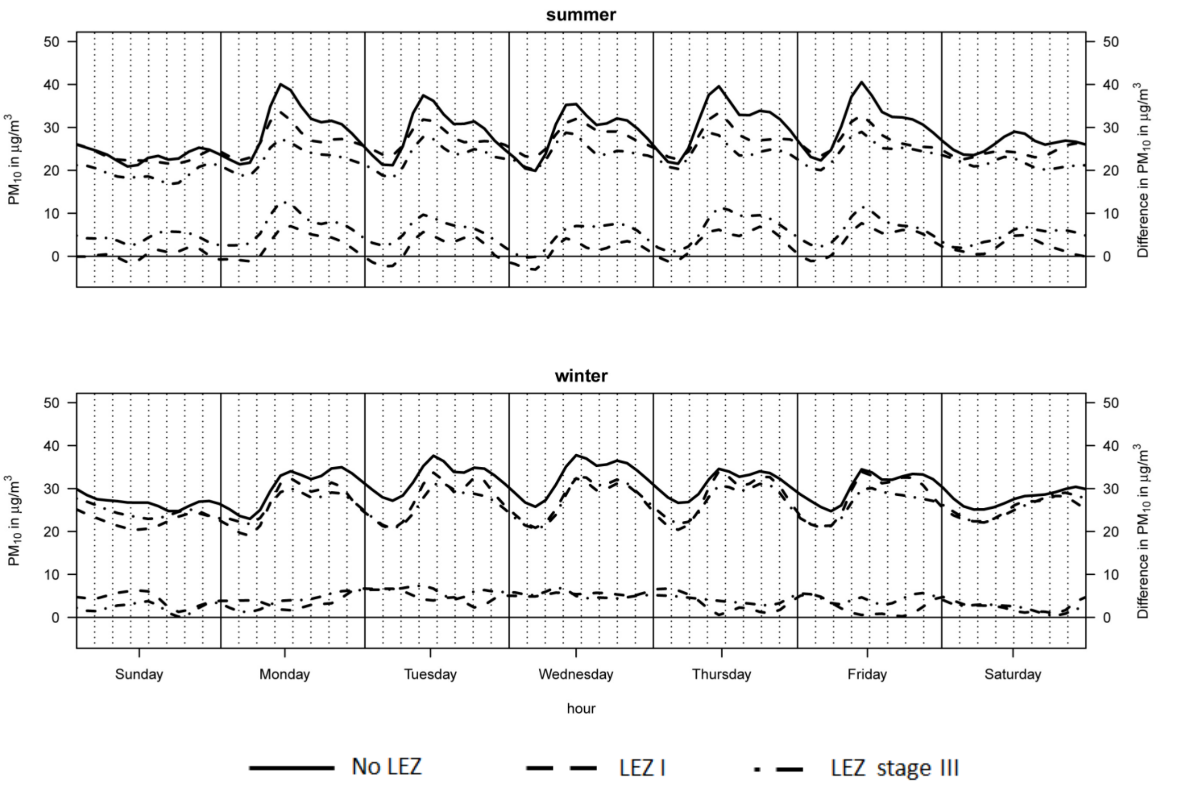



**Figure 31. Modeled hourly concentrations of $PM_{10}$ at Frankfurter Allee adjusted for $PM_{10}$ at the reference station, wind direction and public holidays. Differences in $PM_{10}$ concentrations are shown on the right axis (positive signifies decrease by LEZ implementation).**

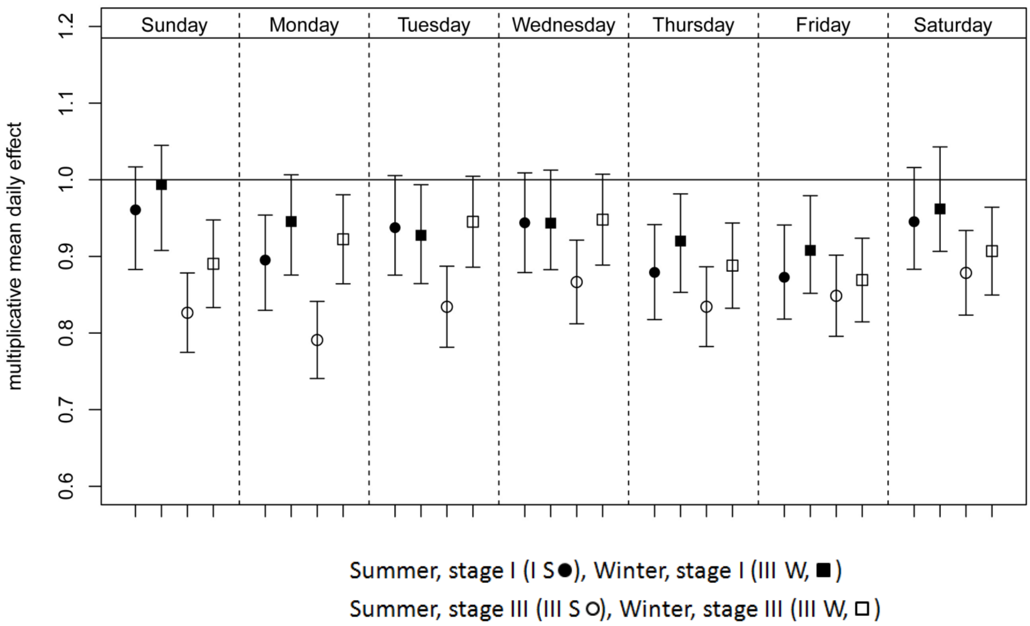



**Figure 32. Daily mean effects of the measures stratified by season and day of the week with 95 % confidence intervals for Frankfurter Allee.**

#### 6.3.3.6 Summary of the results of statistical analysis for Berlin

The changes of $PM_{10}$ concentrations in Berlin in LEZ stage I and LEZ stage III compared with no LEZ period are summarized in Table 11. Season-specific effects of stage I and III of the LEZ, the confidence interval, and the p-values were given. Significant reductions in $PM_{10}$ concentrations were observed at both sites, in both summer and winter season, and for both LEZ I and LEZ III stages. The effects in reducing $PM_{10}$ concentrations were mostly <10% for LEZ stage I, and >10% in LEZ stage III. The strongest reduction was observed in summer of LEZ I stage (-19.0%) at Schildhornstr. The weakest reduction (-5.8%) was observed at Frankfurter Allee for the period 1 (LEZ I) in winter season.

**Table 11. Change of $PM_{10}$ concentration[a] in Berlin in period 1 (LEZ I) and period 2 (LEZ III) compared to period period 0 (no LEZ) at at Schildhornstr and Frankfurter Allee.**

| Measurement site | Summer | | | Winter | | |
|---|---|---|---|---|---|---|
| Period 1 (LEZ I) | | | | | | |
| | Effect | Confidence interval | p-value | Effect | Confidence interval | p-value |
| Schildhornstr. | **-6.9%** | (-10.4%, -3.4%) | <0.001 | **-12.7%** | (-16.0%, -9.4%) | <0.001 |
| Frankfurter Alle | **-8.1%** | (-10.8%, -5.4%) | <0.001 | **-5.8%** | (-8.5%, -3.0%) | <0.001 |
| Period 2 (LEZ III) | | | | | | |
| Schildhornstr. | **-19.0%** | (-21.7%, -16.3%) | <0.001 | **-13.3%** | (-16.2%, -10.4%) | <0.001 |
| Frankfurter Alle | **-16.0%** | (-18.2%, -13.9%) | <0.001 | **-9.1%** | (-11.4%, -6.7%) | <0.001 |

[a] *adjusted for* $PM_{10}$ *concentration at the reference station, wind direction, day of the week, time of the day and public holidays.*

## 6.4 Conclusions from statistical analysis for Augsburg, Munich, Berlin

In this analysis we evaluated the effectiveness of LEZ on the reduction of $PM_{10}$ mass concentration in the ambient air by monitoring data for three German cities: Augsburg, Munich and Berlin. In Munich also a second measure (truck transit ban through the city area) was implemented shortly before the LEZ became effective. The analysis of the routinely collected $PM_{10}$ mass concentrations data was done by a semi-parametric regression model and adjusted for following confounder: $PM_{10}$ levels at the reference site (located in regional background), wind direction, public holidays, day of the week and time of the day. The confounders were selected by a priori consideration. Because of the seasonal variation in $PM_{10}$ concentrations, which could make the magnitude of the effect estimates different for summer and winter season, we modelled the $PM_{10}$ concentrations for both seasons separately by introducing of an indicator function for “summer without measures”, “summer with measures”, “winter without measures” and “winter with measures”.

The $PM_{10}$ data were routinely collected at monitoring site operated by the local network. The analysis was conducted for three monitoring sites in Augsburg, three sites in Munich and two sites in Berlin (including traffic and urban background site).

The results for Augsburg were not consistent both regarding the differences between summer and winter seasons as well as between traffic sites and urban background site. A decrease of $PM_{10}$ concentrations in Augsburg was observed only in the summer season and only at traffic monitoring sites, not in the urban background. For the winter season a clear trend of $PM_{10}$ levels was not detected. Whereas at KS site and BP site (traffic and background site, respectively) an increase of $PM_{10}$ concentrations was observed, the $PM_{10}$ levels at KP site (traffic site) decreased after the implementation of the first LEZ stage (LEZ I) and remained stable after the implementation of the second LEZ stage (LEZ II). Therefore we concluded that the impact of the LEZ in Augsburg on the $PM_{10}$ levels is rather weak, if any.

On the contrary a clear reduction of $PM_{10}$ levels was observed in Munich and Berlin after implementation of LEZ in those cities. The magnitude of the reduction was larger for Berlin, the decrease of $PM_{10}$ concentration in Berlin range between -6 and -19% depending on the monitoring site and the active stage of the LEZ.

In general, the decrease after the implementation of further LEZ stages (LEZ II in Munich and LEZ III in Berlin) was larger compared to the reduction after implementation of LEZ I. In Berlin the reduction almost doubled after the implementation of LEZ III in the summer season, whereas the difference between the LEZ I and LEZ III in winter season was less pronounced. Also in Munich the decrease after the second stage of the LEZ became effective was more pronounced (up to -15% at a traffic site Landshuter Allee) as for LEZ I only.

We observed also clear seasonal differences regarding the magnitude of the effect. The reduction of $PM_{10}$ levels was in general more pronounced for the summer season compared to the winter season (in Munich at all sites, in Berlin with one exception: Schildhornstrasse after implementation of LEZ I). In winter, additional particle sources (such as domestic heating, wood combustion, re-suspended dust due to the application of road salt for deicing) contribute significantly to the $PM_{10}$ mass concentrations in the ambient air. In addition, the generation of secondary aerosols such as nitrate or sulphate is more intensive in winter. Consequently, exhaust particles represent a smaller fraction of the fine particles in winter than in summer. This could be the reason that measures regulating only the exhaust particles could became less effective in the winter period. In addition, adverse meteorological conditions leading to increased $PM_{10}$ levels from local mobile as well as stationary sources are occurring more frequently in the winter season. Our analysis suggests that in such episodes the influence of the implemented measures regulating the car exhaust is limited and that air quality is dominated by other unaffected mobile and stationary sources.

## 6.5 Assessment of LEZ impact on AQ in Augsburg based on modeling

### 6.5.1 Introduction

Augsburg is a mid-sized (270.000 inhabitants) city located in a populated part of Germany, only some 60 km northwest of Munich which has >1.5 million inhabitants. We should expect the air pollution levels in the centre of Augsburg to be influenced by a rather important regional background. A large regional contribution may imply inter-annual variations in meteorological conditions – e.g. in dominant wind direction – that affect the absolute levels of the Augsburg urban background from one year to another. Interannual variations in the air pollution levels of the air masses arriving to Augsburg may hide the pollution reductions gained from the LEZ enforcement. Moreover, interannual variations in the meteorological conditions also influence the dispersion of locally emitted pollutants, giving different air pollution concentrations in response to local emissions. We have therefore two scientific questions that we aim to answer in this modelling study;

- What is the expected size of the reduction of the urban background air pollution levels due to the introduction of LEZ in Augsburg and is it possible to see this reduction in the measured urban background levels?
- Is there a significant variability in meteorology that affects the measured annual mean urban background air pollution concentration in Augsburg?

The here described modelling analysis of the LEZ impact focuses the urban background pollution levels, this since the model used outputs roof level concentrations. The analysis of street level measurements close to traffic is performed with another method, see preceding section 6.3.

### 6.5.2 Methods

In order to estimate the effects of the LEZ stage 1 introduction in 2009, and its further enforcement with stage 2 introduced in 2011, we have performed an analysis based on existing monitoring data, meteorological information and existing emission inventories, using the Airviro modelling system (http://www.airviro.smhi.se). It is a web based system for air quality management that comprises measurements and measurement collection, emission inventories, and dispersion and wind modelling tools. The system is applied in multiple countries and can be used for different scales, from urban to large scale applications.

We have modelled the contribution of local Augsburg emissions to the urban background concentrations of $PM_{10}$ and $NO_x$ registered at the monitoring stations. We have not modelled the regional contribution, which thus must be determined either from other assessments or estimated indirectly from monitored levels.

In the following sections we describe the observational data we use, the input emissions to the modelling system as well as the modelling system in itself. Finally we describe the case studies conducted to answer the questions.

### 6.5.3 Meteorological and air quality data

Meteorological data from 2008 to 2011 were extracted from Augsburg airport, operated by the Deutcher Wetterdienst (DWD; http://www.dwd.de). The Airviro model system uses vertical temperature gradient close to the ground as an important variable to calculate stability and scaling parameters. Since this information is not available, we used wind velocity, temperature, global radiation and cloudiness to calculate hourly values for the vertical temperature gradient (Berkowicz and Prahm, 1982a and 1982b). Global radiation were not available at the Augsburg airport, instead recordings at HA monitoring site were used.

The monitor data to be compared with model output should as much as possible represent the urban background concentrations in Augsburg. The monitoring stations in Augsburg that are classified as urban background stations are (see map in Figure 33):

- The ***LfU*** site is located on the premises of Bavarian Environment Agency (LfU), ca 4 km south of the city centre.

- The ***HA*** (Hochschule Augsburg) site measurement location is placed on the campus of the University of Applied Sciences Augsburg which is approximately 1 km to the south-east of the city center. Within a radius of 100 m it is surrounded by campus buildings, a tram depot and a small company. The nearest main road is in the north-east at a distance of 100 m. Within a radius of approximately 200 m the monitoring site is almost completely surrounded by residential areas, apart from a small park located in north-western direction (Pitz et al., 2008).

- The Bourgesplatz (***BP***) is located in a small park. As a major road (and a tram line) is located about 70 meters north of BP site, this site is considered as urban background site (though maybe influenced by nearby traffic) (Gu et al., 2013).

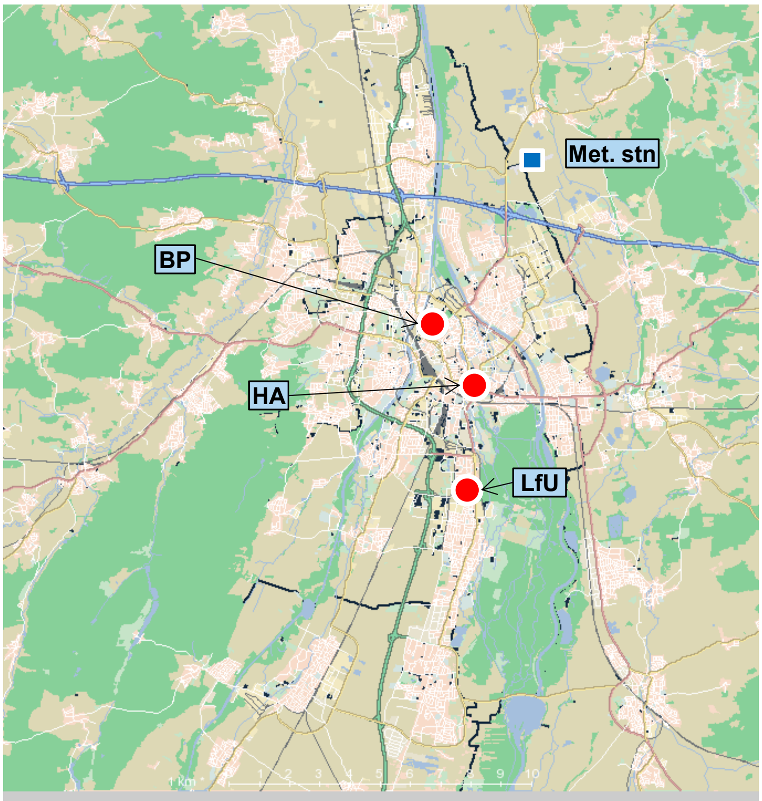



**Figure 33. Location of meteorological station and urban background monitoring stations.**

### 6.5.4 Emissions

Our goal is to describe the emissions for 2008 (baseline) and then analyze the effect of the LEZ introduction 2009 and 2011. The 2008 baseline inventory is based on two data sources:

- Traffic on Augsburg road network for gasoline cars PKW/B, diesel cars PKW/D and commercial traffic 2009; with percentage of heavy duty vehicles (given in a shape file, simulated by a traffic model)
- Other PM and $NO_x$ emissions, delivered on a 2x2 $km^2$ grid, for the year 2004 separated in
  - offroad traffic (201)
  - licensed industrial (202)
  - non-licensed combustion sources (203)
  - non-licensed other sources (204)
  - other (205), remaining sources

  (from EKATBY, 2008, or "Fortschreibung des Emissionskatasters Bayern für das Jahr 2004", Stuttgart University)

We have thus a baseline emission inventory, aimed to describe the year 2008, where traffic data are from 2009 and other emission data from 2004. Since our purpose is to determine the relative impact of LEZ on air pollution levels in Augsburg, we will assume that this baseline inventory gives a reasonably appropriate level of total emission for 2008. The only thing we will change while simulating the coming years 2009-2011-2013 are traffic emission factors as required by the LEZ introduction. All other emissions will be kept constant. This assumption is likely not correct since traffic volume often grows by time; which would lead to an overestimation of the reduction effect of the LEZ implementation. Restrictions like LEZ may also lead to less traffic within the LEZ area and more traffic outside, when some old and high-emitting vehicles are forced to stay outside. We have not had access to such traffic scenarios, rather we have assumed that all traffic in the Augsburg area, not only the ones driving inside the LEZ area, are following the LEZ emission standard requirements.

The traffic information is given directly on the road network, which gives a very high spatial resolution. The EKATBY emissions inventory, representing the year 2004, have a spatial resolution of 2x2 $km^2$ and are described as emission grids released close to ground. The 202 Licensed industrial source showed very high emissions in one particular grid cell, representing a major industry whose emissions in reality is coming out from an elevated point source (stack). We therefore used more detailed information on the location and height of individual stacks: 142 point sources for $PM_{10}$ and 78 point sources for $NO_x$. These point sources replaced the 202 grid emissions in the modelling to be presented below.

The simulated traffic volumes included weekly and diurnal variations, here taken from the Swedish national modelling system SIMAIR (Gidhagen et al., 2009). Figure 34 shows the diurnal variations for a typical city road, for light duty vehicles and heavy duty vehicles.

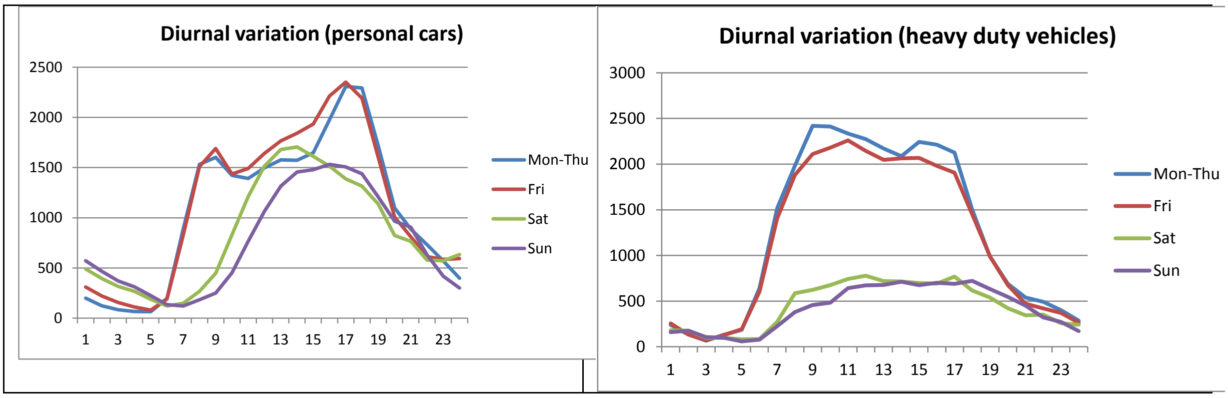



**Figure 34. Diurnal variations of traffic volume for light duty (left) and heavy duty (right) vehicles. Unit: relative variation.**

Time variations were also introduced for emission sources 203, 205 and 205 (EKATBY, 2008), see Figure 35.

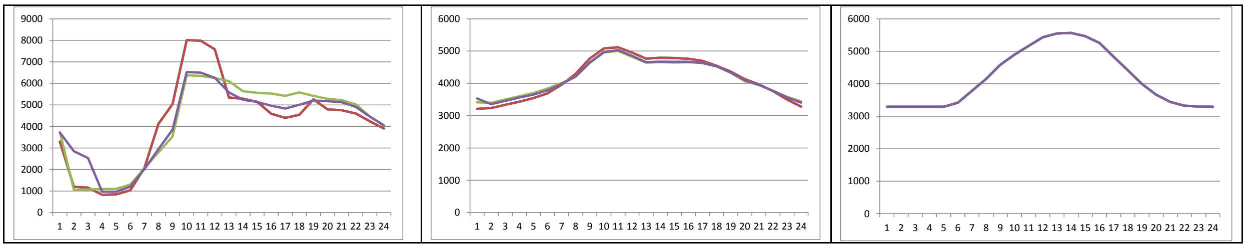



**Figure 35. Diurnal variation (relative) for emission source 203 (left), 204 (middle) and 205 (right).**

The traffic activity was combined with emission factors taken from EMPLAN (2012). Emission factors depending on Euroclass were weighted by the fractions of vehicles of each Euroclass for each year. For the baseline the emission factors were based on Euroclass information on registered vehicles, and for the projections it is based on 100% compliance (EMPLAN, 2012). For the commercial traffic, the data was divided between light duty vehicles on gasoline (LNF/B), regional busses on diesel (RBus/D), city busses on diesel (L/Bus/D) and heavy duty traffic on diesel (SNF/D). To calculate the total emission fraction for all commercial traffic, the traffic counts[1] of these vehicle types were used for the 17 roads included in EMPLAN (2012), and a weighted emission fraction was calculated to represent the emissions of commercial traffic in all of Augsburg. These emission factors were calculated for $NO_x$ and $PM_{10}$ (exhaust emissions only), for the years 2008, 2009, 2011 and 2013. The resulting emission factors are presented in Table 12.

**Table 12. Emission factors (mg/veh,km) used for light duty vehicles (LDV) and heavy duty vehicles (HDV) for the simulated emission years in the evaluation of the low emission zone (LEZ).**

| | | 2008 | 2009 | 2011 | 2013 |
|---|---|---|---|---|---|
| LEZ | | | Stage 1 July 1st | Stage 2 January 1st | |
| **$PM_{10}$** | LDV | 15.26 | 11.86 | 7.63 | 3.44 |
| | HDV | 170.34 | 91.10 | 65.67 | 36.73 |
| **$NO_x$** | LDV | 344.68 | 327.79 | 298.76 | 263.45 |
| | HDV | 7014.01 | 6357.12 | 2030.44 | 1685.50 |

Uncertainties in methods

- The $NO_x$ emission factors in Table 12, which are based on HBEFA emission factors, may be underestimated in comparison with real world conditions (Carslow et al., 2011).
- The emission factors do not include stop-and-go. How this affects the emissions will depend on how the traffic evolves inside and outside LEZ, with more traffic and congestion or with less traffic and more free flow driving.

[1] Based on live traffic counts in 2007 on the 17 roads.

- The LEZ implementation effect on the vehicle fleet is based on 100% compliance and not on registered vehicle fleet information. This may result in overestimation of the effect of the LEZ.
- The LEZ implementation is assumed to affect all vehicles in Augsburg, i.e. traffic emissions outside LEZ has also been reduced equally as inside. It is probable that the compliance is lower outside the zone than inside it. This will likely result in an overestimation of the LEZ effect.
- Traffic activity is held constant at the level of 2009. Traffic volume in general has a tendency to increase with time, at the same time as a LEZ implementation has a potential to reduce the traffic inside and increase it outside. Thus the constant traffic assumption can contribute both to over- and underestimation of the LEZ effect on the urban background levels.

The summed effect of these uncertainties is likely that our analysis gives an overestimated LEZ effect on the urban background concentrations of $NO_x$ and $PM_{10}$ in Augsburg.

#### 6.5.5 Modelling

A Gaussian dispersion model, part of the AIRVIRO system, was used to conduct $PM_{10}$ and $NO_x$ modelling on 250x250 $m^2$ spatial resolution. The emissions inventory was formed using the data and methodology described above and directly accessible through the system. The digital map, land use distribution and topography were taken from the Open-Street web page. The modelling domain and grid resolution is displayed in Figure 36.

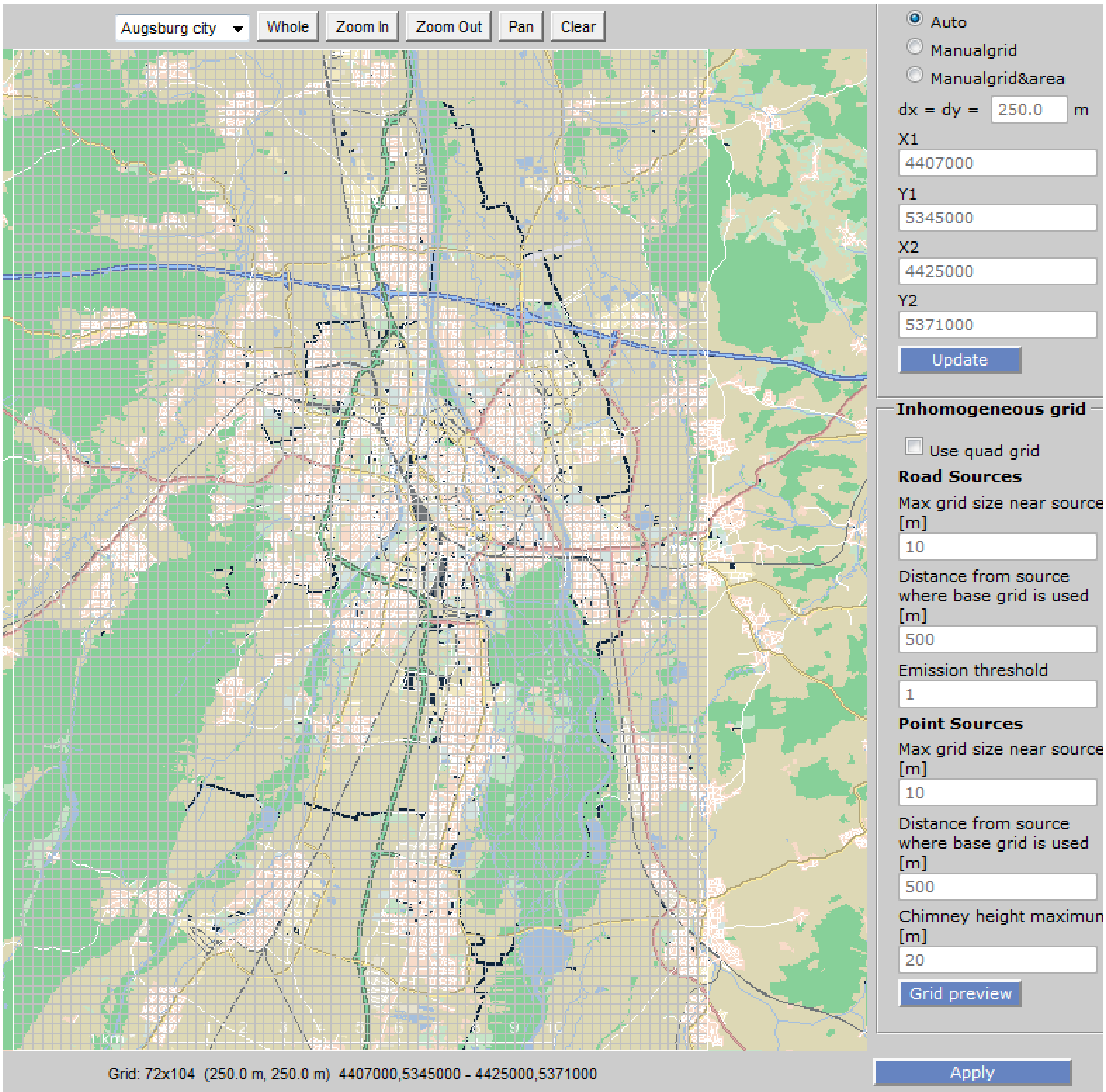



**Figure 36. Modelling domain with spatial resolution.**

Our task is to give answers to the scientific questions given in the introduction to this section:

- What is the expected size of the reduction of the urban background air pollution levels due to the introduction of LEZ in Augsburg and is it possible to see this reduction in the measured urban background levels?
- Is there significant variability in meteorology that affects the measured annual mean urban background air pollution concentration in Augsburg?

The LEZ were introduced in two steps during the years 2009 to 2011, i.e. we should expect the lower emissions from transport to be reflected in changes of the pollution levels. Our approach has been an assessment based on the following analysis, for which we will present results:

- Magnitude and spatial distribution of emissions from different source sectors (baseline year 2008)
- Monitored annual averages of $PM_{10}$ and $NO_2$ levels (2008-2009-2010)
- Simulated impact of local emissions before LEZ (2008)
- Simulated impact of local traffic emission change with LEZ (2009-2011-2013)
- Other factors influencing the urban background in Augsburg:
  - meteorological variability (2008-2009-2010-2011)
  - regional background contribution and variability

#### 6.5.6 Magnitude and spatial distribution of emissions from different source sectors

The main characteristics of the baseline emission database, named 2008 to show that it represents conditions prior to LEZ, is presented in Figure 37 ($NO_x$) and Figure 38 ($PM_{10}$).

For $NO_x$, road traffic and industrial emissions are comparable in magnitude, however the latter is strongly concentrated to the northeastern part of Augsburg. As commented before it was necessary to describe the industrial emissions as point sources with specified stack heights.

For $PM_{10}$ the largest source is the non-licensed combustion (86 tonnes/year), which includes small consumers (such as the military) and households (including residential wood combustion). The traffic source (35 tonnes/year), the licensed industry (35 tonnes/year) and the non-licensed industry (32 tonnes/year) are all similar in size while the off-road traffic is smaller (15 tonnes/year). For traffic, only combustion emissions are included; there is no estimation of e.g. road dust and other non-combustion PM emissions caused by road traffic. If model simulations are to be used to determine the local contribution to PM exposure, road dust should be included. However, the reason behind only exhaust emissions being included in this assessment is that this source is the only relevant for the demonstration of the LEZ impact. LEZ only regulates combustion PM and not the non-exhaust contribution.

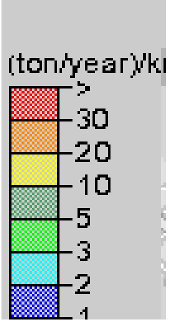
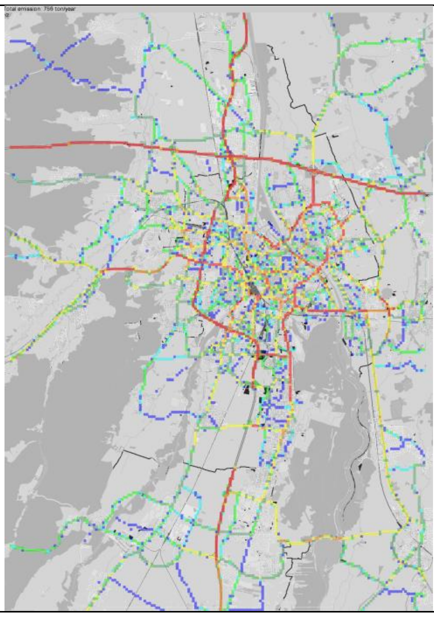
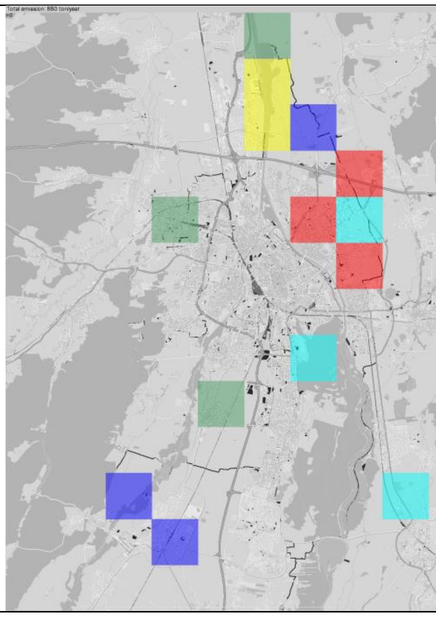
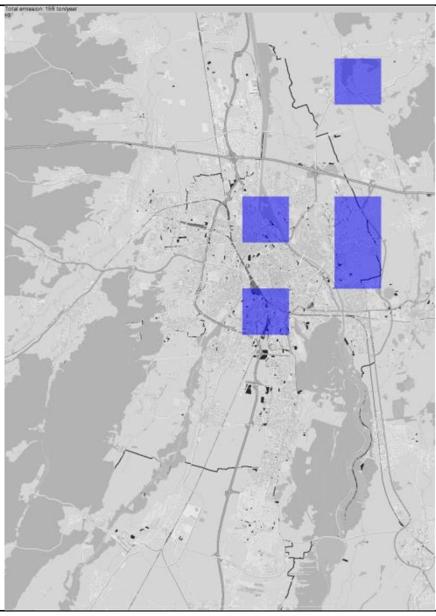
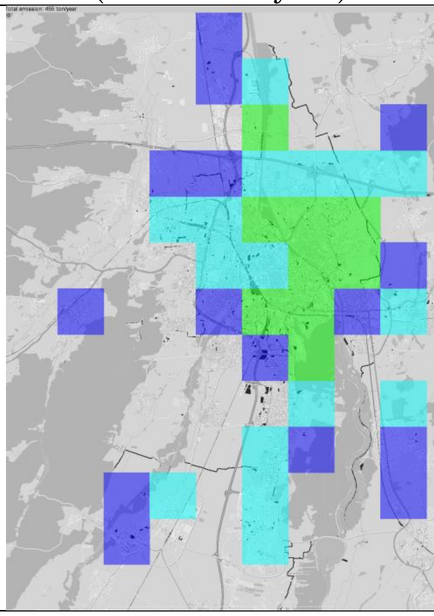
| | | |
|---|---|---|
|  |  |  |
| | Road traffic (888 tonnes/year) | Licensed Industry 202 (880 tonnes/year) |
| |  |  |
| | Off-road traffic[2] 201 (156 tonnes/year) | Non-licensed combustion[3] 203 (455 tonnes/year) |

**Figure 37. $NO_x$ spatial distribution and total emission over model domain for the year 2008 (before LEZ) separated in sectors. All sectors were emitted at ground level except sector 202, which was emitted as elevated point sources.**

[2] Includes railway traffic, shipping and aviation.

[3] Includes small consumers (such as the military) and households (including residential wood combustion)

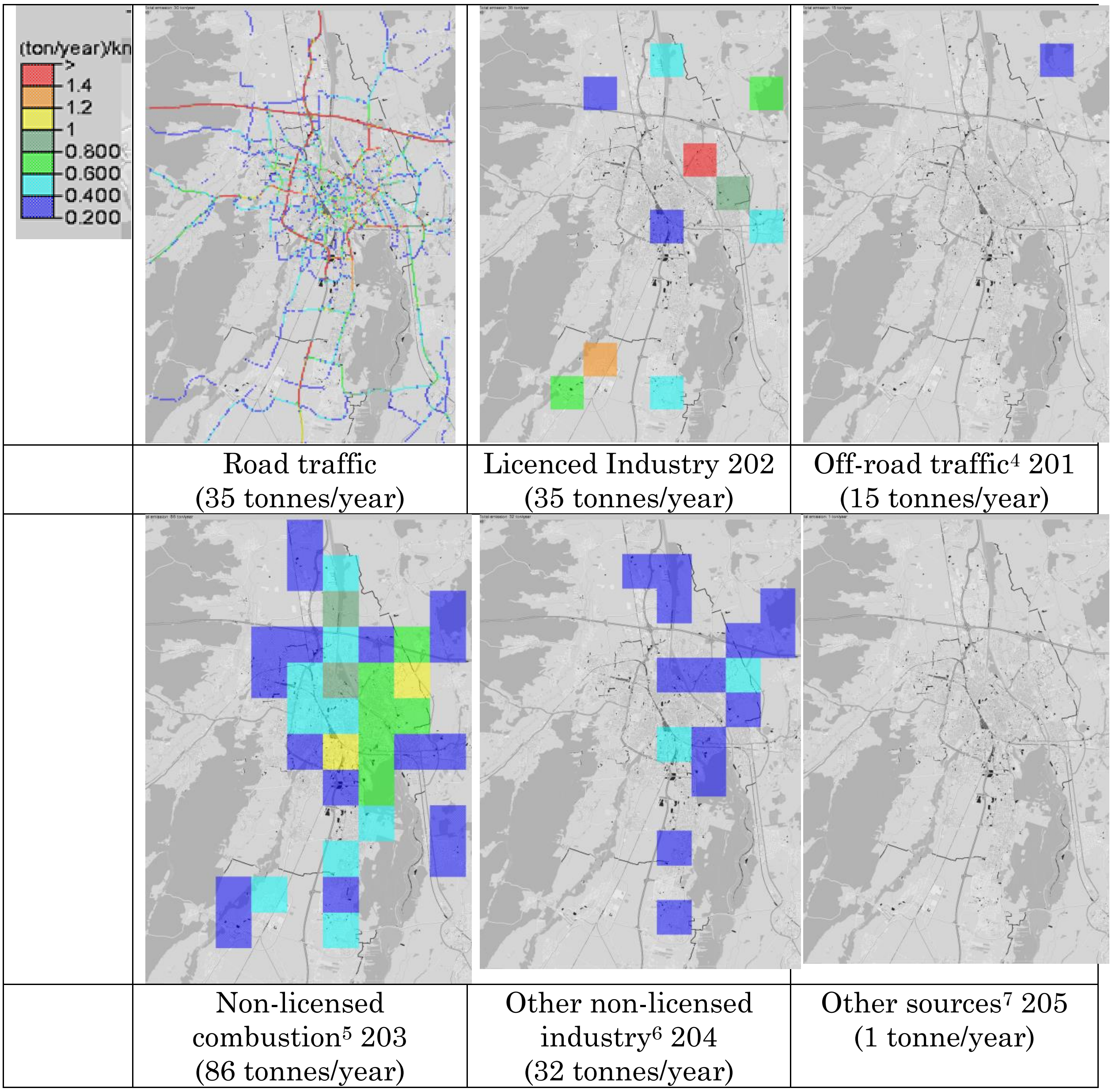



**Figure 38. $PM_{10}$ spatial distribution and total emission over model domain for the year 2008 (before LEZ) separated in sectors. All sectors were emitted at ground level except sector 202, which was emitted as elevated point sources.**

[4] Includes railway traffic, shipping and aviation.
[5] Includes small consumers (such as the military) and households (including residential wood combustion)
[6] Includes emissions from dusty goods, from agricultural activities, petrol stations etc.
[7] Includes e.g. agriculture, landfills, waste water treatment

### 6.5.7 Monitored annual averages of $PM_{10}$ and $NO_2$ levels 2008-2009-2010

Comparing three years of monitoring data (see Figure 39), we notice minor differences between the years 2008 (before the LEZ) and 2010 (LEZ first step). There is no signal of reduced pollution levels after the LEZ according to these data, on the contrary there is a tendency of slightly higher levels of the measured urban background for the year 2010 as compared to 2008. Besides the urban background sites (BP, HA and LfU), two street level measurement sites (KP and KS) are also included in the figure. The LEZ impact should be more significant on those street level stations, but the analysis of them is left out here (see instead the statistical analysis of Section 6.3).

From the three years of monitored data, we see interannual variations in urban background of almost 10 µg/m³ for $NO_2$ and about 3 µg/m³ for $PM_{10}$.

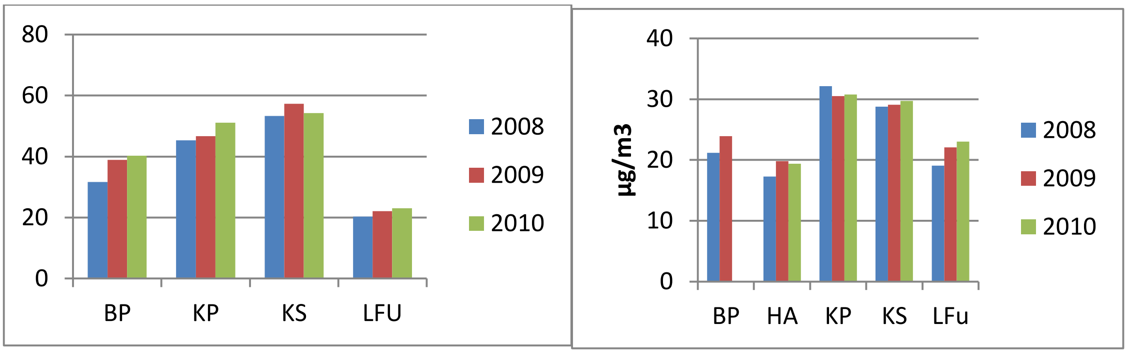



**Figure 39. Monitored annual average concentrations of $NO_2$ (left) and $PM_{10}$ (right) at four stations in Augsburg.**

### 6.5.8 Simulated impact of local emissions before LEZ (2008)

Simulations have been performed for $PM_{10}$ and $NO_x$ for the baseline year 2008[8]. Figure 40 shows that local traffic contributes to less than 0.5 µg/m³ and the impact on urban background of all local $PM_{10}$ emissions is 2 µg/m³ or lower in the LEZ area, as compared to the measured total $PM_{10}$ of around 20 µg/m³ at the urban background sites.

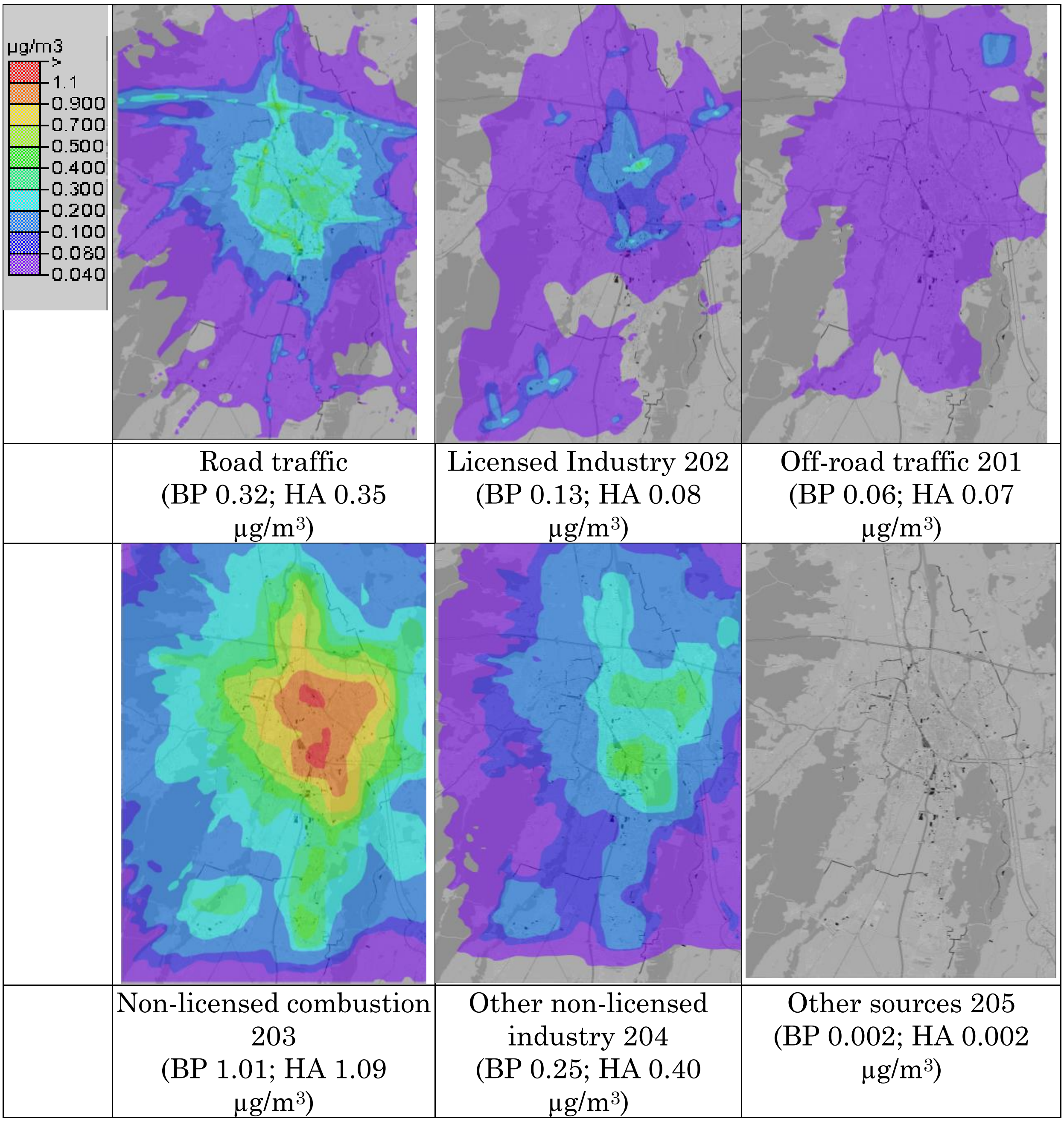



**Figure 40. Simulated annual average $PM_{10}$ concentration contributions from different sectors/sources for the baseline year 2008 (before LEZ). Total concentrations from local sources: BP: 1.76 µg/m³, HA: 1.99 µg/m³ and LfU: 0.96 µg/m³.**

[8] In this simulation meteorology of the year 2008 was used.

Figure 41 shows that emissions from traffic create local $NO_x$ contributions to the urban background annual mean above 10 µg/m³ (marked red in map, max 17 µg/m³ during 2008).

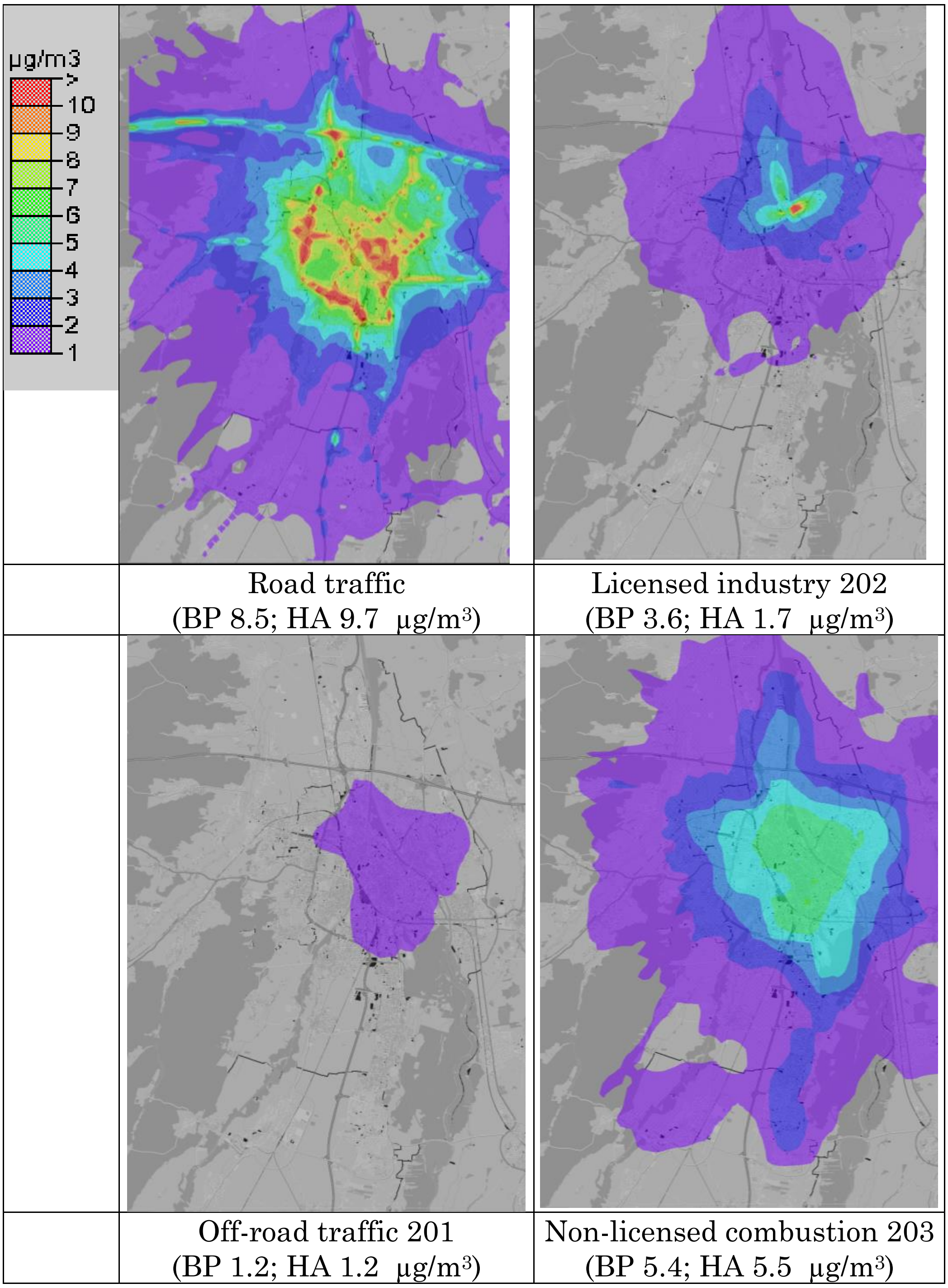



**Figure 41. Simulated annual average $NO_x$ concentration contributions from different sectors/sources for baseline year 2008 (before LEZ). Total concentrations from local sources: BP: 18.6 µg/m³, HA: 18.1 µg/m³ and LfU: 8.7 µg/m³.**

Figure 42 shows the annually averaged simulated local contributions at urban background monitoring stations BP, HA and LfU. The simulated levels with emissions from all local sources can be compared to the monitored during 2008:

Station BP:

- $PM_{10}$: Simulated 1.8 µg/m$^3$ compared to monitored 21.2 µg/m$^3$
- $NO_x$: Simulated 18.6 µg/m$^3$ compared to monitored $NO_2$ 31.6 µg/m$^3$

Station HA:

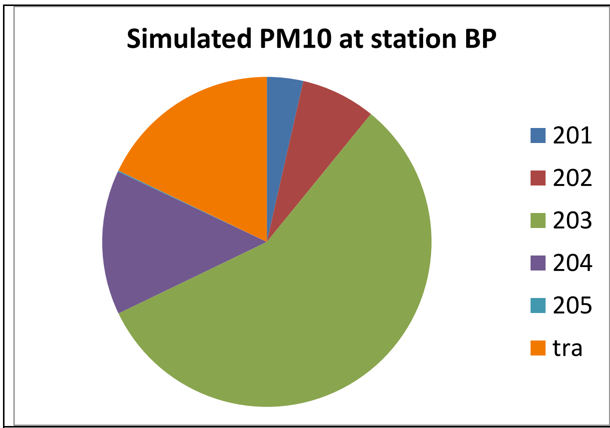
- $PM_{10}$: Simulated 2.0 µg/m$^3$ compared to monitored 17.3 µg/m$^3$
- $NO_x$: Simulated 18.1 µg/m$^3$

Station LfU:

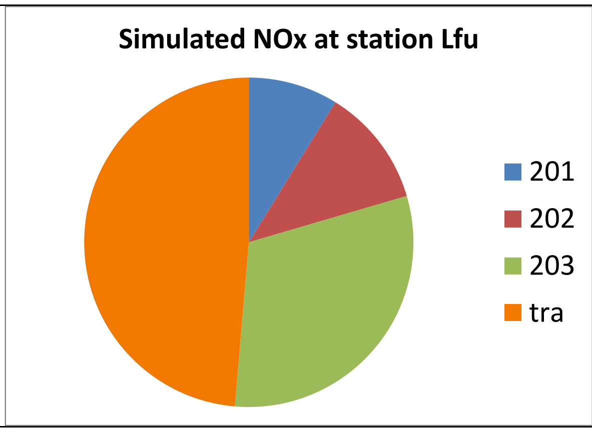
- $PM_{10}$: Simulated 1.0 µg/m$^3$ compared to monitored 19.1 µg/m$^3$
- $NO_x$: Simulated 8.7 µg/m$^3$ compared to monitored $NO_2$ 20.4 µg/m$^3$

We can see that HA has a larger contribution from local sources, however a lower $PM_{10}$ concentration is registered at HA compared to BP. This indicates that BP is located closer to the traffic sources and that HA is a more well-defined urban background station (this is also in line with the description of the stations). Adding a road dust contribution of similar size as the exhaust contribution (0.35), we can estimate a $PM_{10}$ regional contribution from sources outside the modelling domain of about 15 µg/m$^3$.

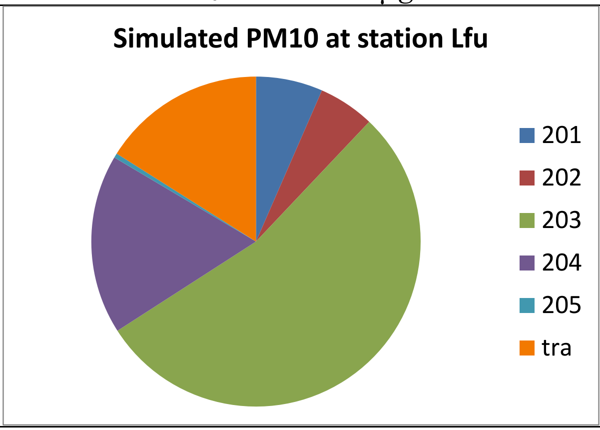
The corresponding estimation of a regional background for $NO_2$ is more uncertain, since we simulate $NO_x$, whereas $NO_2$ (not $NO_x$) is available from the BP and LfU stations. If we assume that $NO_x$ is 10-30% higher than $NO_2$ in the urban background (based on Stockholm urban background air data), then we could expect a $NO_x$ regional background of about 15 µg/m$^3$ (if comparing with LfU) or 20 µg/m$^3$ (if comparing with BP). It is likely that the estimation based on LfU is representative of the regional background, and that local sources with large impact on the BP site are under-estimated.



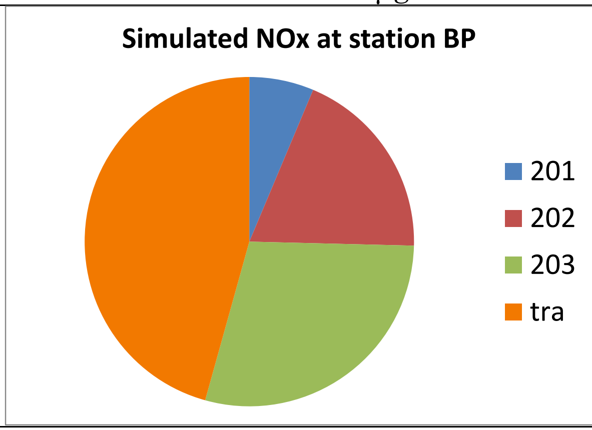

$PM_{10}$ total: 1.8 µg/m³



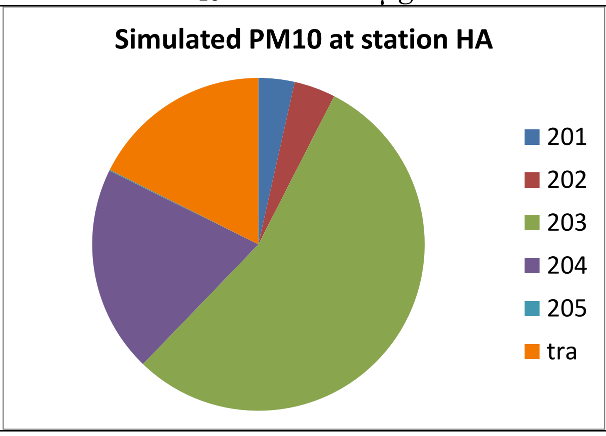

$NO_x$ total: 18.6 µg/m³



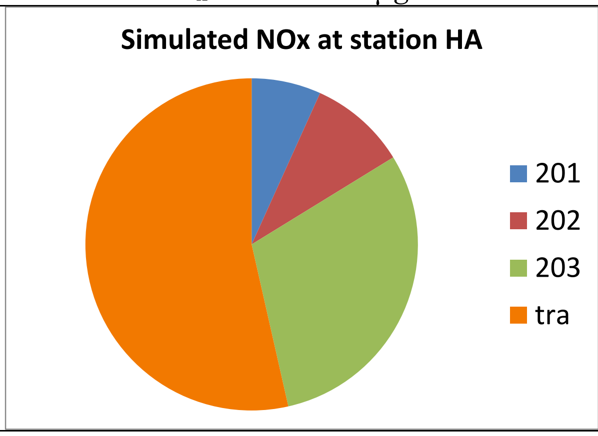

$PM_{10}$ total: 2.0 µg/m³




$NO_x$ total: 18.1 µg/m³




$PM_{10}$ total: 1.0 µg/m³




$NO_x$ total: 8.1 µg/m³

**Figure 42. Simulated contributions to $PM_{10}$ at the stations BP, HA and Lfu from Augsburg sources during 2008.**

The following graphs show the time variation of monitored and simulated concentrations at urban background stations and averaged for working days Monday to Thursday. Figure 43 shows that simulated $PM_{10}$ from local sources is of the order of 10 times lower than monitored $PM_{10}$ in urban background sites. For $NO_x$ (Figure 44) the simulated values are about half of measured $NO_2$ concentrations during workdays. If we assume that $NO_x$ is 10-30% higher than

$NO_2$ in the urban background[9], we can assume local $NO_x$ contributions to be approximately one third of the total $NO_x$.

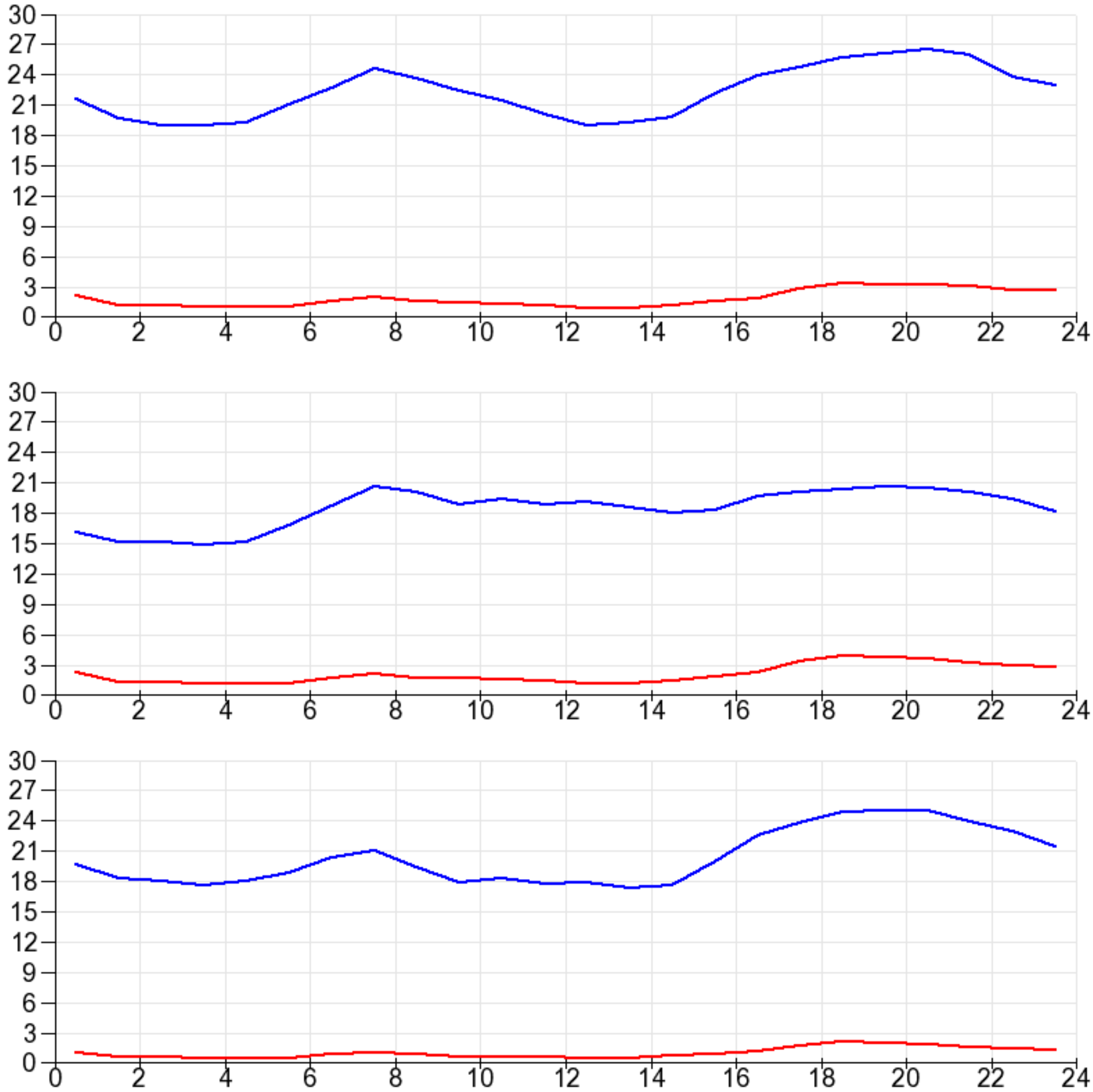



**Figure 43. Diurnal variation of monitored (blue) and simulated (red) $PM_{10}$ during workdays Mon-Thu at the three urban background stations: BP (top), HA (middle) and LfU (bottom). Emissions and meteorology are from baseline year 2008. Unit: µg $m^{-3}$.**

[9] This is based on measurements in the Stockholm urban background air, since there are no available $NO_x$ measurements in Augsburg.

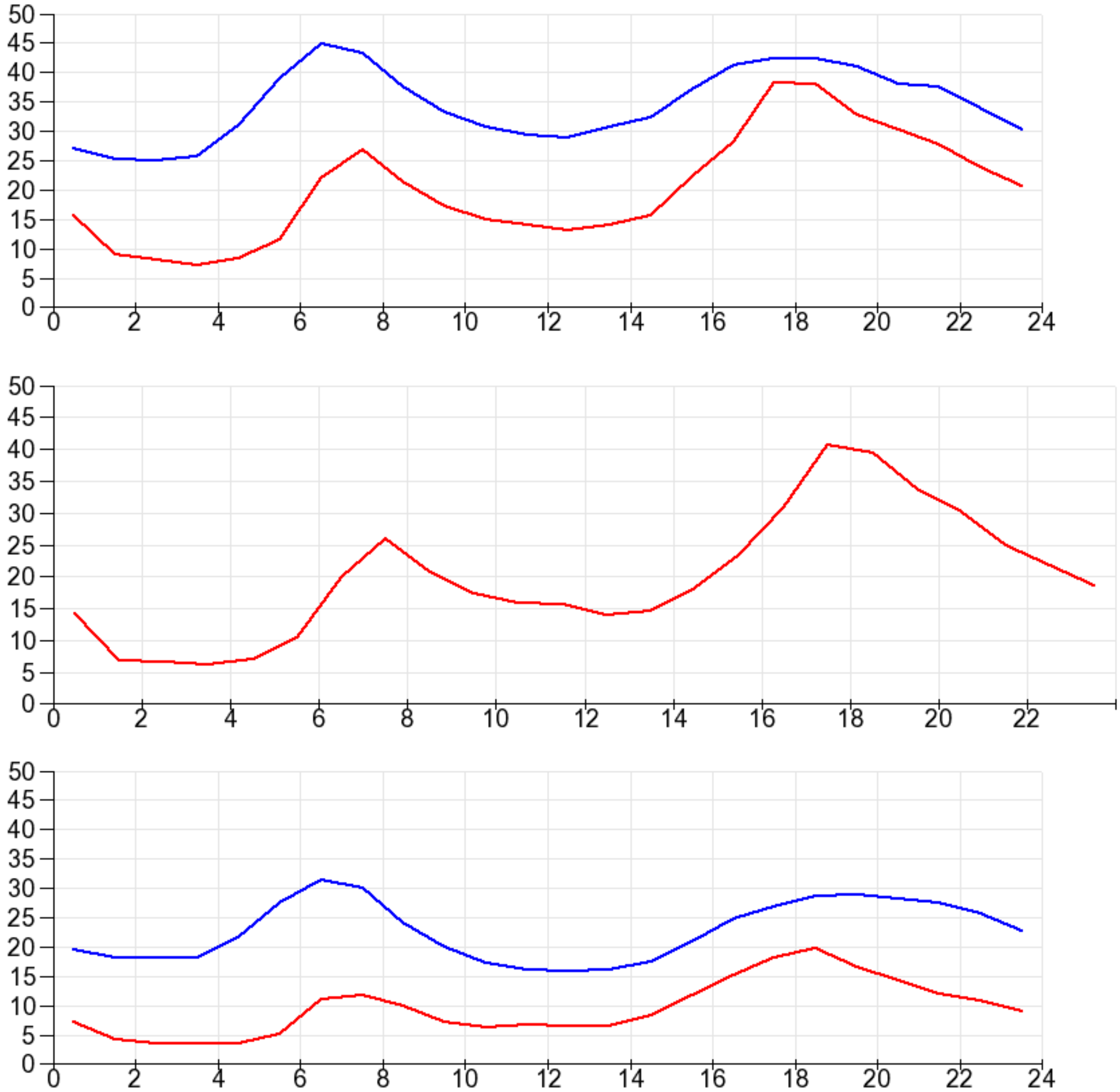



**Figure 44. Diurnal variation of monitored $NO_2$ (blue) and simulated $NO_x$ (red) during workdays Mon-Thu at the three urban background stations: BP (top), HA (middle) and LfU (bottom). Emissions and meteorology are from baseline year 2008. Unit: µg m-3.**

Monitored $PM_{10}$ levels show small morning and evening peaks, rising 10-20% above the nighttime concentrations. The $NO_2$ variations from nighttime levels to morning and afternoon peaks are much larger, rising some 100%. Simulated $PM_{10}$ levels are, as commented earlier, only about 10% of monitored levels, but show a similar daily pattern. Simulated $NO_x$ levels have a higher afternoon peak, while monitored $NO_2$ levels have a more slightly more pronounced morning peak. It is likely that the daily variation in traffic emissions with a high evening peak in traffic volume (Figure 35)[10] is not representative in Augsburg.

[10] This variation is based on Swedish conditions, taken from the Airviro modelling system.

### 6.5.9 Simulated impact of local traffic emission change with LEZ (2008-2009-2011-2013)

To determine the expected benefit of LEZ we have also created traffic emissions for the years 2009 (stage 1), 2011 (stage 2) and 2013 (stage 3[11]) using the emission factors that comply with LEZ requirements. We compare the concentrations resulting from the stage-wise LEZ introduction to those of the baseline year, 2008, which were described and compared to measurements in the previous section. All local emissions, besides the traffic emissions, are considered to remain on a constant level, i.e. as in the baseline 2008 emission database, to isolate the impact of changes in traffic emissions due to the LEZ. The emission reductions displayed in Figure 45 are then resulting from the LEZ introduction which affects the road traffic emissions.

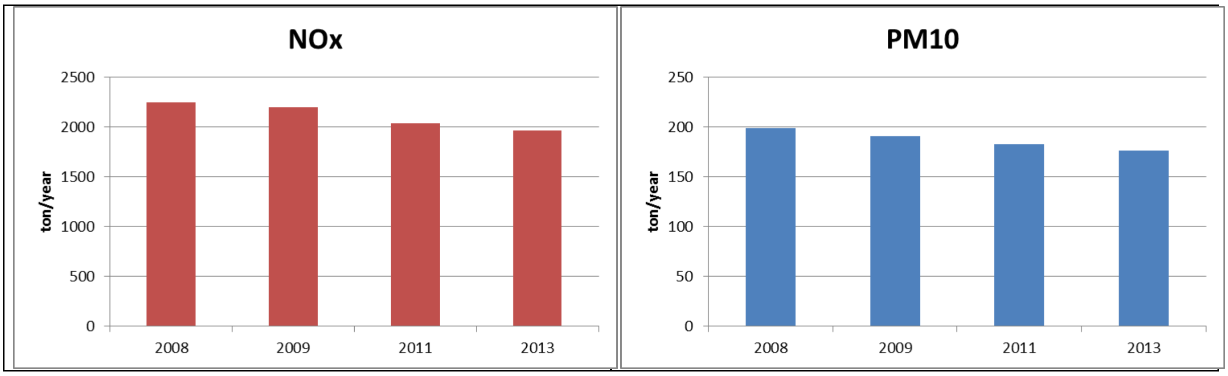
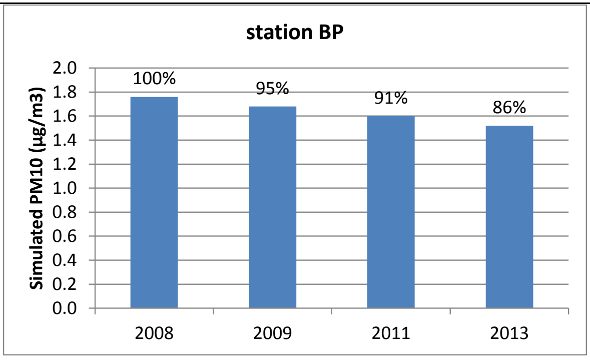


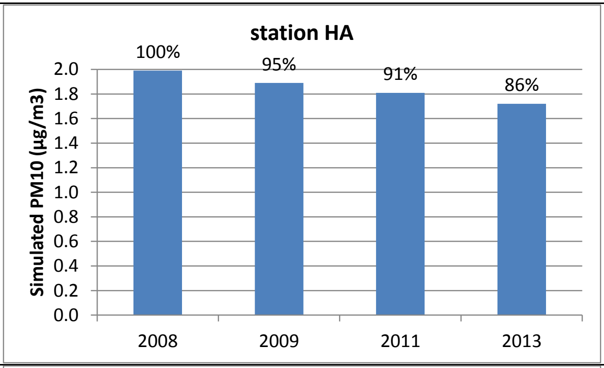

**Figure 45. Evolution of total local emissions for $NO_x$ (left) and $PM_{10}$ (right) through the stage-wise introduction of LEZ (meteorology for baseline year 2008 used for all simulations, only changing traffic emissions).**

In order to eliminate the effect of meteorological variability, we have used the same meteorological year (2008) for all four simulations of 2008-2009-2011-2013 traffic emission scenarios. The resulting concentrations with enforced LEZs are shown in Figure 46 for $PM_{10}$ and in Figure 47 for $NO_x$. The net reduction in annual averaged urban background concentration levels expected, due to the stage 3 (year 2013) LEZ implementation, will be up to 0.3 µg/m$^3$ for $PM_{10}$ and up to 4 µg/m$^3$ for $NO_x$. These reductions should be compared to total monitored levels in city centre of about 20 µg/m$^3$ for $PM_{10}$ and about 40 µg/m$^3$ for $NO_2$ ($NO_x$ levels are slightly ~10-30% higher[12]). This means that we will have difficulties to see effects of LEZ by comparing variations in annual mean monitoring data. Further, inter-annual variability in meteorological conditions and regional background levels are bound to impact the urban background levels. In the next section we analyse these factors.

[11] The enforcement of stage 3 has been delayed. Thus, 2013, should be seen as a pure future scenario.

[12] This is based on measurements in the Stockholm urban background air, since there are no available $NO_x$ measurements in Augsburg.



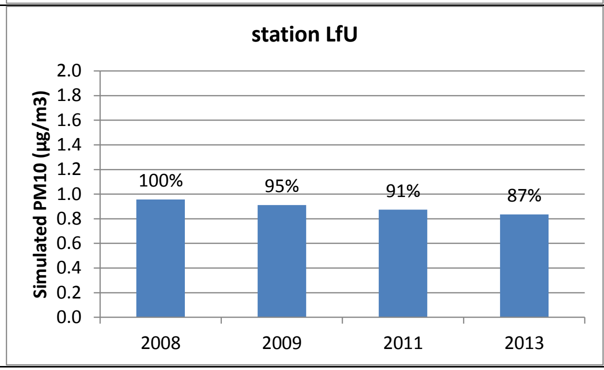
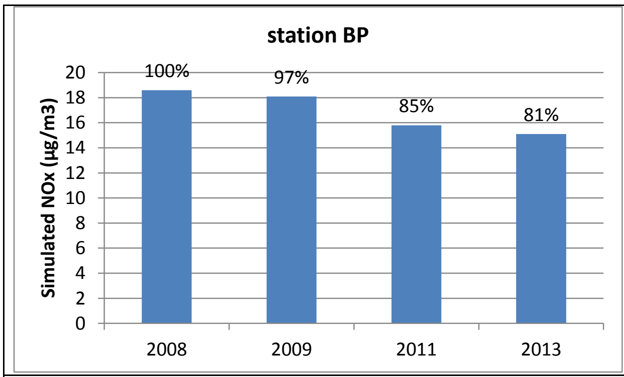
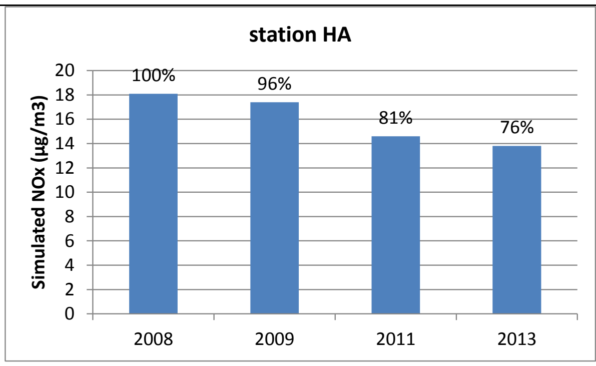




**Figure 46. Simulated impact of total local $PM_{10}$ emissions at three urban background stations. Reduction in $PM_{10}$ levels are only due to changes in vehicle emissions due to LEZ (meteorology from 2008 used in all four simulations).**



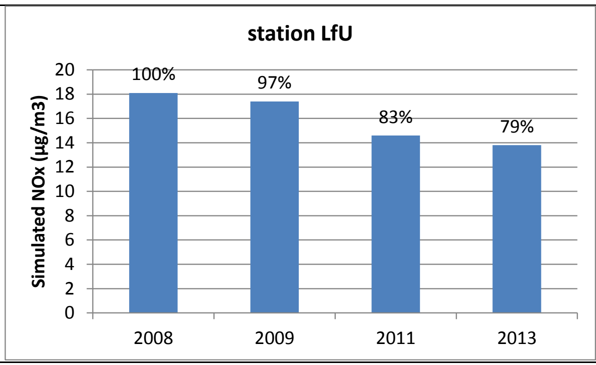




**Figure 47. Simulated impact of total local $NO_x$ emissions at three urban background stations. Reduction in $NO_x$ levels are only due to changes in vehicle emissions due to LEZ (meteorology from 2008 used in all four simulations).**

### 6.5.10 Impact of meteorological year-to-year variation on the urban background in Augsburg

The model simulations have shown that the LEZ in Augsburg contributes to reductions of up to 4 µg/m$^3$ in $NO_x$ and 0.3 µg/m$^3$ in $PM_{10}$. We will now quantify the variability caused by the meteorological forcing for the years 2008, 2009, 2010 and 2011.

First we investigate wind roses for the years to see if there are notable differences (Figure 48). For all years there is a dominance of wind from WSW and secondarily from ENE, with smaller differences in the frequency of winds from other directions.

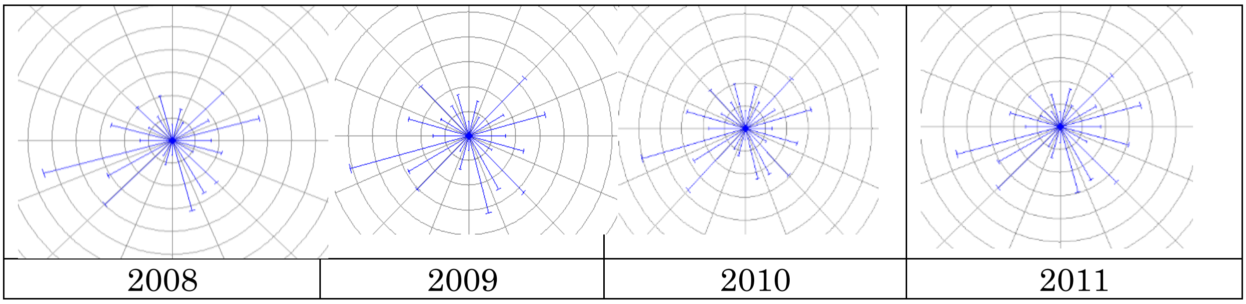



Figure 48. Wind roses from meteorological station at the Augsburg airport.

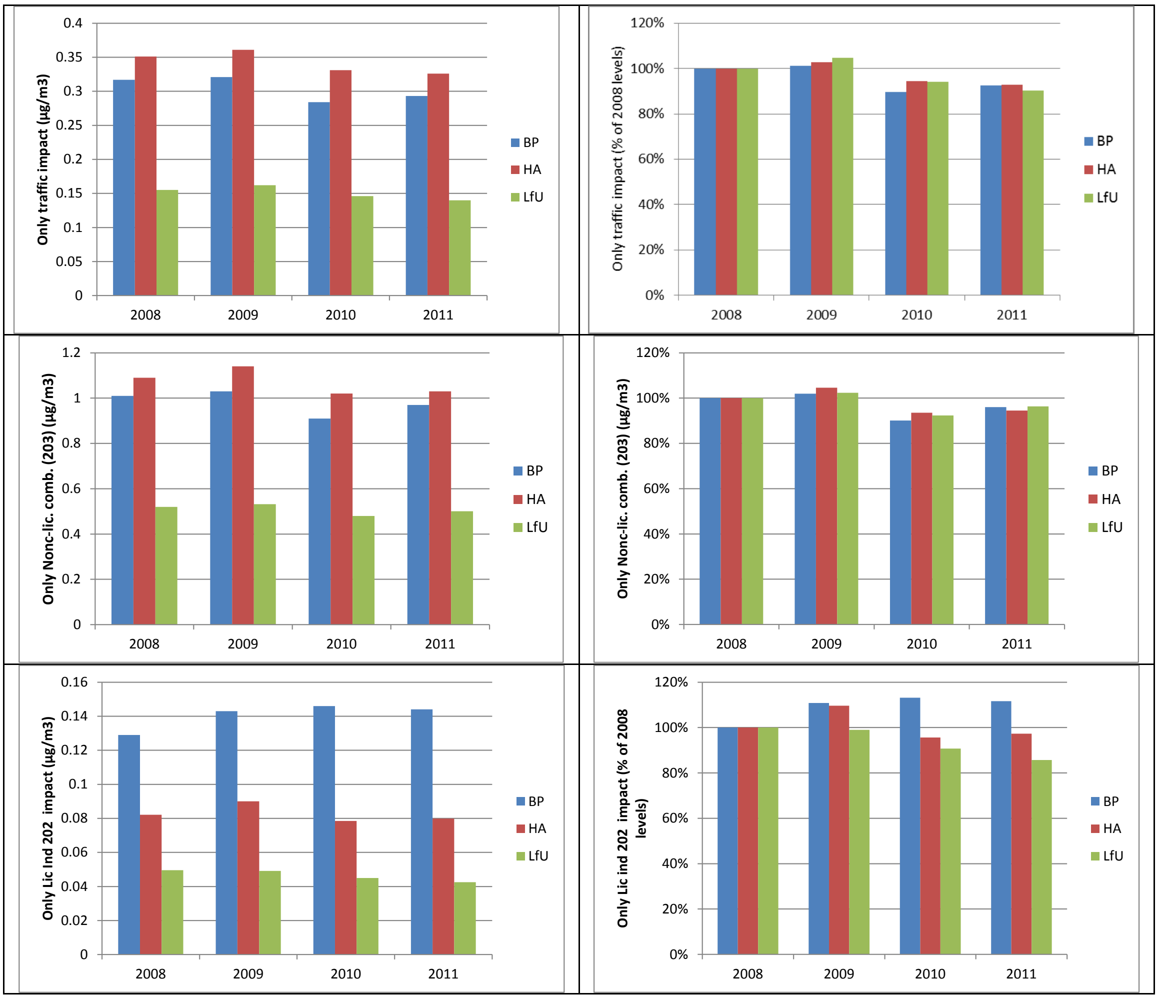



Figure 49. Simulated contributions of $PM_{10}$ to BP, HA and LfU monitoring stations due to traffic sources (top), non-licensed combustion (203; middle) and elevated industrial sources (202; bottom) for the meteorological years 2008-2009-2010-2011. The only difference between the years is the meteorological conditions. The diagrams to the right are expressed as percentage of the 2008 annual levels.

A more quantitative approach is to run the model for the four years, but using the same local emissions (the same as in the baseline, 2008). The difference in final concentrations between the years will then reflect the meteorological interannual variability. The three diagrams to the left in Figure 49 show the simulated impact of three types of emissions, first from traffic emitted close to ground, second the largest PM source non-licensed combustion which is also emitted close to ground and third from industrial emissions through elevated stacks. The diagrams to the right show how the annual averages in percentage of the 2008 levels.

We can see that there is an interannual variation of ± 10-15%. For ground-level sources there is a tendency for 2008-2009 to have had slightly higher urban background levels as compared to 2010-2011. For elevated point sources the pattern is different at one station (BP), where 2008 is lower than 2009-2010-2011. This is likely due to wind directions (BP is close to the largest point source), while the general trend of 2008-2009 being higher than 2010-2011 is more coupled to wind speed and dilution conditions.

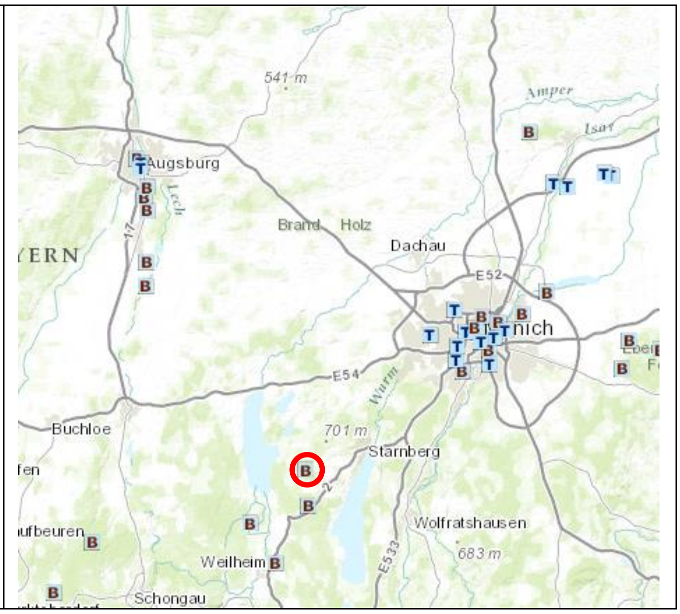
The interannual variations in meteorology will lead to variations in the urban background contribution of $PM_{10}$ and $NO_x$ of about 10%. This means year-to-year variations in the impact of local sources of around 0.2 μg/m$^3$ for $PM_{10}$ and around 2 μg/m$^3$ for $NO_x$. These differences are of the same magnitude as the expected reductions due to LEZ. We will therefore look at the tendencies of monitored values.

Monitored $NO_2$ and $PM_{10}$ (see Figure 39) at the three urban background stations show no sign of lower levels for 2010, instead 2008 stands out with the lowest $PM_{10}$ and $NO_2$ levels. This indicates that the monitored urban background year-to-year variation neither responds to LEZ emission reductions, nor do they reflect meteorological variability in the dispersion and dilution of local emissions.

### 6.5.11 Regional background concentrations

There is one rural monitoring station DEBY109 located ~55 km south of Augsburg and ~35 km southwest of Munich, see map in Figure 50.

**Figure 50. Monitoring station DEBY109 (marked with a red ring), located at about 700 m over sea level.**



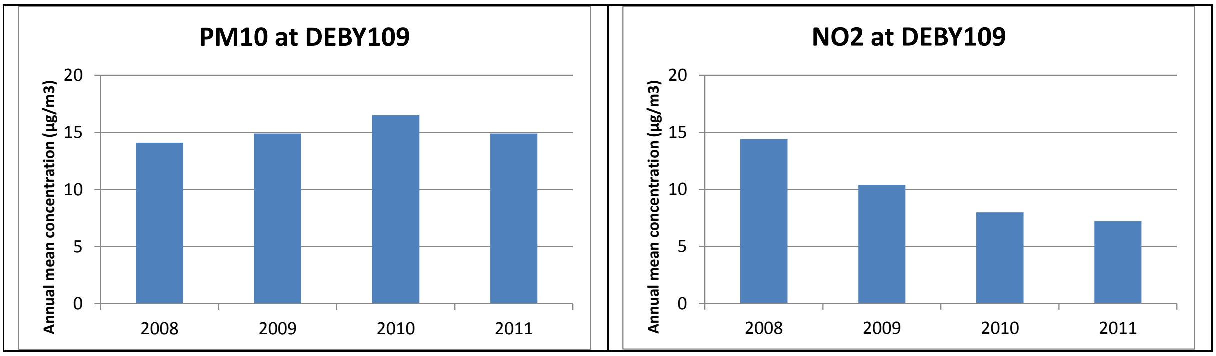




**Figure 51. Monitored annual mean concentration levels at station DEBY109.**

During the period 2008 to 2011 there are interannual differences of about 2.5 µg/m$^3$ for $PM_{10}$ and about 7 µg/m$^3$ for $NO_2$ at the DEBY 109 rural background station (Figure 51). For $PM_{10}$ we can see a similar variation in the Augsburg urban background as in DEBY109, moreover a regional background level of the same magnitude (15 µg/m$^3$) as we determined from the gap between monitored values at Augsburg urban background sites and the simulated impact of local sources.

For $NO_2$ however, there is no similarity between the very strong decrease at DEBY109 and those at the urban background sites in Augsburg. It seems very unlikely that the regional $NO_2$ contribution arriving to Augsburg has a variation like the one in DEBY109. Thus for $PM_{10}$, DEBY109 can be used for estimating the Augsburg regional background concentration, while it does not seem to be representative for $NO_2$.

### 6.5.12 Summary of inter annual variations in simulated and monitored pollution levels

Our analysis has shown that the year-to-year variations during 2008-2009-2010 of monitored $PM_{10}$ and $NO_2$ in the urban background of Augsburg cannot be explained by neither LEZ emission reductions, nor by changed dispersion conditions due to interannual meteorological variations. However, there is an indication that the long-range contribution of $PM_{10}$ can explain the monitored variations in Augsburg urban background air. For $NO_2$ such conclusions are not possible, since the only rural information available show a strong decrease that is not seen in Augsburg background air. An explanation why DEBY109 can serve as a regional background station to Augsburg for $PM_{10}$ but not for $NO_2$ is that $PM_{10}$ has a much larger regional contribution which makes the spatial gradients smaller over this part of Germany. $NO_2$ is affected much more by local emissions along major roads and in each village, implying stronger spatial gradients.

In Table 13 and 14 we summarize the concentrations levels and trends obtained in measurements and simulations. We discuss for $PM_{10}$ the urban background measured in station HA, since this station seems to be more representative (less affected by traffic emissions close to the station) as compared to BP. As $NO_2$ is not measured in HA, we present both BP and LfU, the latter situated in the periphery of Augsburg.

The idea is not to explain in detail the monitored variability in the Augsburg urban background, but rather to give a picture on how difficult it is to document the positive effects of LEZ. This partly because the traffic in Augsburg is, especially for $PM_{10}$, only a minor source to the urban background concentrations (lower than 20 % - see Gu et al., 2013) and variations in other factors may easily obscure the LEZ effect. We have seen that meteorological variability can produce variations in the impact of local sources of the same magnitude as the expected LEZ reductions (after its full stage 3 implementation), however it does not seems to be a dominating factor for what happens in the urban background between 2008 and up to 2010. Instead it may be variations in the regional background that obscure minor LEZ reductions. We can see from Table 13 that monitored rural $PM_{10}$ concentrations have the absolute levels and year-to-year variations that together with the simulated $PM_{10}$ (possibly also with some added impact of non-exhaust emissions from road traffic) sum up to levels similar to the monitored urban background. Although DEBY109 is not a perfect rural background to Augsburg, it indicates the magnitude of a large-scale $PM_{10}$ distribution over this part of Germany.

Table 14 shows that for $NO_2$ during 2008 we have a relatively good similarity between the DEBY109 rural background plus the local impact on the one hand and the monitored urban background on the other. However, the $NO_x$ concentrations in DEBY109 show a strong decrease during the four years shown in Table 15 which does not at all correspond to what is registered in stations BP and LfU. The stronger gradients in $NO_2$ distribution, as compared to $PM_{10}$, invalidate the use of DEBY109 as an indicator of the magnitude of the $NO_2$ concentrations in air masses arriving to Augsburg.

**Table 13. Interannual variations in annual means of monitored and simulated $PM_{10}$ contributions. Urban background refer to station HA.**

| Annual mean $PM_{10}$ ($\mu g/m^3$) at station HA | 2008 | 2009 | 2010 | 2011 | 2013 |
|---|---|---|---|---|---|
| Monitored rural background (DEBY109) | 14.1 | 14.9 | 16.5 | 14.9 | |
| Monitored urban background | 17.3 | 19.8 | 19.4 | | |
| Simulated total urban background due to traffic emission change[13] | 2.0 | 1.9 | | 1.8 | 1.7 |
| Simulated traffic contribution due to traffic emission change[14] | 0.35 | 0.25 | | 0.17 | 0.08 |
| Simulated meteorological variability in traffic contribution[15] (ground source) | 0.35 | 0.36 | 0.33 | 0.33 | |
| Simulated meteorological variability in non-permit combustion sources 203 (ground source)[15] | 1.09 | 1.14 | 1.02 | 1.03 | |
| Simulated meteorological variability in license industry contribution 202 (elevated source)[15] | 0.08 | 0.09 | 0.08 | 0.08 | |

[13] Only traffic emissions were altered between the years. Meteorology and other emissions were held constant.
[14] Only traffic emissions were altered between the years. Meteorology were held constant.
[15] All emissions were held constant at the baseline level. Only meteorology varied between the years.

**Table 14. Annual mean of measured $NO_2$ and simulated $NO_x$ contributions at the BP and LfU stations for different sources and scenarios.**

| **Annual mean $NO_2$ and $NO_x$ ($\mu g/m^3$) at station BP** | **2008** | **2009** | **2010** | **2011** | **2013** |
|---|---|---|---|---|---|
| Monitored rural background $NO_2$ (DEBY109) | 14.4 | 10.4 | 8.0 | 7.2 | |
| Monitored urban background $NO_2$ | 31.6 | 38.9 | 40.3 | | |
| Simulated total urban background $NO_x$ due to traffic emission change[16] | 18.6 | 18.1 | | 15.8 | 15.1 |
| Simulated traffic contribution $NO_x$ due to traffic emission change[17] | 8.5 | 8.0 | | 5.7 | 5.0 |
| **Annual mean $NO_2$ and $NO_x$ ($\mu g/m^3$) at station LfU** | **2008** | **2009** | **2010** | **2011** | **2013** |
| Monitored rural background $NO_2$ (DEBY109) | 14.4 | 10.4 | 8.0 | 7.2 | |
| Monitored urban background $NO_2$ | 22.3 | 22.1 | 23.0 | | |
| Simulated total urban background $NO_x$ due to traffic emission change[18] | 8.68 | 8.4 | | 7.2 | 6.9 |
| Simulated traffic contribution $NO_x$ due to traffic emission change[19] | 4.2 | 4.0 | | 2.7 | 2.4 |

### 6.5.13 Conclusions from modelling of the LEZ effect in Augsburg

We have analysed the $PM_{10}$ and $NO_2/NO_x$ urban background concentrations by running a dispersion model with available emission data covering the Augsburg model domain. By comparing with monitor data we conclude:

- $PM_{10}$: Local contribution from sources inside the modelling area contributes to about 10% (2 $\mu g/m^3$) of monitored levels (about 17-18 $\mu g/m^3$) in the city centre (monitor station HA). A regional contribution from sources outside Augsburg of about 15 $\mu g/m^3$ (varying by 3 $\mu g/m^3$ during the period 2008-2010) explains the other 90% of the monitored levels in the urban background.
- $NO_2/NO_x$: Local contribution from sources inside the modelling area contributes to about one third to half (19 $\mu g/m^3$ of $NO_x$ at station BP) of monitored levels (30-35 $\mu g/m^3$ of $NO_2$ estimated to correspond to around 40 $\mu g/m^3$ of $NO_x$) in the city center. The regional background contribution is estimated to be about 15 µgm3.
- The expected reductions of the urban background levels due to LEZ enforcement of less vehicular emissions, are up to 0.3 $\mu g/m^3$ for $PM_{10}$ and 4 $\mu g/m^3$ for $NO_2$. According to the planned LEZ implementation, this will

[16] Only traffic emissions were altered between the years. Meteorology and other emissions were held constant.
[17] Only traffic emissions were altered between the years. Meteorology were held constant.
[18] Only traffic emissions were altered between the years. Meteorology and other emissions were held constant.
[19] Only traffic emissions were altered between the years. Meteorology were held constant.

occur around 2013 (however stage 3 is still not implemented). This will give an expected LEZ effect of 1-2% lower $PM_{10}$ concentrations in the Augsburg center. Corresponding LEZ effect on $NO_2$ urban background concentrations is slightly under 10%.

The monitored $PM_{10}$ and $NO_2$ levels show an increase so that the levels in 2010 are some 10% higher as compared to baseline year 2008. This means that some other factor is obscuring the expected impact of LEZ emission reductions in the measurements.

We have analyzed:

a) meteorological variability leading to different dispersion and dilution of local emissions: these cannot explain the monitored variations, as they will contribute to lower values 2010 as compared to 2008.

b) variability in the regional contribution: monitored concentrations in the rural station DEBY109 show a similar increase and can explain why Augsburg urban background $PM_{10}$ rises from 2008 to 2010. However, the DEBY109 $NO_2$ values do not seem to be representative for the air masses arriving to Augsburg ($NO_2$ is not so dominated by regional contributions as $PM_{10}$ and is varying more over the rural area outside Augsburg and Munich).

There are various other possibilities that can modify and obscure the expected reduction of urban background levels due to LEZ, e.g. errors and/or changes in non-traffic emissions, increases in traffic volume etc. These factors have not been investigated in detail, as this would require both an updated and detailed emission inventory and specific monitoring campaigns.

### 6.5.14 Comparison of LEZ impact on AQ in Augsburg based on statistical analysis of monitoring data and AQ modelling

LEZ effects on air quality in Augsburg have been evaluated separately by both statistical analysis of monitoring data and an air quality dispersion model. There are substantial differences between the two methods used, which gives complementary information illustrating the LEZ impact. For a better understanding of the results obtained from two methods, a discussion follows.

#### 6.5.14.1 Methods

Statistical analysis is a “receptor-oriented” method starting from ambient $PM_{10}$ concentrations and ending with the changes of $PM_{10}$ due to LEZ. It utilizes real monitored $PM_{10}$ mass concentrations across the city, and controls the confounders that will potentially influence the $PM_{10}$ mass concentrations. After adjusting these confounders, the net change of $PM_{10}$ is considered to be related to the effects of LEZ. There are many factors affecting the ambient $PM_{10}$ concentrations, including the changes of local sources, photochemical reactions, meteorological conditions, and long-range transport etc, among which the LEZ only contribute to the change of local sources. In order to easily eliminate factors other than LEZ that will influence the $PM_{10}$ concentration, assumptions are made in the statistical analysis, which include 1) $PM_{10}$ at a selected reference site outside LEZ is mainly driven by the regional background $PM_{10}$, and not so much affected by LEZ; 2) $PM_{10}$ at traffic and urban background sites within LEZ are affected by the introduction of LEZ and also by the regional background of $PM_{10}$. By using the reference $PM_{10}$ in the statistical model, most of the undesirable factors influencing the $PM_{10}$ are considered to be adjusted.

One limitation of this strategy is, that also long-term changes of $PM_{10}$, which could not be explained by the included confounder variables, especially by the $PM_{10}$ levels of the reference station (for example changes in heating habits or in the local structure of particle sources), are completely attributed to the LEZ effect.

The assessment made with a dispersion model starts from emission inventories and ends with the expected changes in urban background $PM_{10}$ and $NO_x$ due to the requirements that LEZ puts on local traffic emissions. An important assumption made in the analysis was that LEZ did not only affect the traffic emissions inside the defined and very small LEZ area, instead all road traffic in Augsburg was assumed to follow the LEZ emissions regulation.

By comparing model results with measured levels inside and outside Augsburg, the interannual variability caused by meteorological conditions and long range transport have been estimated and compared with the expected LEZ effect.

#### 6.5.14.2 Results

For statistical analysis, no consistent results were found in Augsburg. The effects of LEZ in reducing $PM_{10}$ were observed at traffic sites in summer season, while no effect was observed for urban background site in summer. In winter, effects were only observed at KP site (traffic site), while at KS (traffic site) and BP (urban background site), an increase in $PM_{10}$ was observed. Overall, from the statistical analysis we assume the effect of LEZ in Augsburg is rather small in comparison to other factors that contribute to total $PM_{10}$ levels.

The dispersion modelling approach shows that emissions within Augsburg only contribute to about 10% of monitored $PM_{10}$ levels, while local emissions contribute to about one third to half of monitored $NO_2$ levels. The expected effect of LEZ is 1-2% lower $PM_{10}$ and slightly below 10% lower $NO_2$ in the LEZ urban background. This should be compared to year-to-year variations in meteorological conditions that create variations in local contributions to urban background $PM_{10}$ and $NO_x$ levels of the same magnitude as the expected LEZ impact. For $PM_{10}$ a stronger year-to-year variation was found in the long-range transported air arriving to

Augsburg. The analysis points to the variations in long-range transported $PM_{10}$ as the factor that obscures the LEZ impact in monitored urban background. Due to lack of information on rural $NO_2$ concentrations, it was not possible to draw the same conclusion for this pollutant.

Linking the two assessments: The statistical modelling could not find significant effects of LEZ in the Augsburg urban background $PM_{10}$ data, whereas the dispersion modelling approach indicate possible explanations to this. For $PM_{10}$ local traffic gives only minor contributions to the urban background levels and year-to-year variations in long-range concentrations are dominating over the expected LEZ reductions.

#### 6.5.14.3 Uncertainties

Uncertainties associated with the statistical analysis will come from model assumptions and other unknown/unaccounted factors that affect $PM_{10}$ concentrations. The former could be influence of LEZ on reference $PM_{10}$ concentration. It is likely that the car fleet in the whole city was renewed due to LEZ, thus vehicles driving outside the LEZ emitted less $PM_{10}$ as well. In spite of this, the reference site is located in the background site outside LEZ, and such influence can be small compared with the traffic site inside the LEZ. The latter uncertainty could be other events/sources not known to our best knowledge, but influenced the ambient $PM_{10}$ concentrations.

As usual in air quality modelling, there are larger uncertainties expected in emissions as compared to meteorology and model imperfections. Major uncertainties are found in the existing emission inventory EKATBY 2004, likely of special importance concerning the large emissions of the dominant source non-licensed combustion (203). The fact of not including non-exhaust emissions from road traffic in the model simulations may have reduced the traffic impact with about 50%, however it did not affect the determination of the LEZ effect on urban $PM_{10}$ levels since we kept the traffic intensity constant before-after LEZ introduction. It is to note that if LEZ actually decreases traffic volumes inside the LEZ area, then the non-exhaust part will also decrease and the LEZ impact will be larger (an indirect effect). Despite the simple assumptions made, the model approach was able to illustrate the expected magnitude of urban background contributions from different local sources in and around Augsburg. This helps to explain the difficulty to find significant effects by LEZ in a statistical analysis of total $PM_{10}$ levels in a rather small city such as Augsburg.

# 7 A literature review of LEZ/CCS impact assessments based on monitoring and modelling

There are a number of studies on the air quality impacts of LEZs (Sadler Consultants, 2011). They are either based on air quality measurements, or using air quality dispersion modelling. In this section we will shortly summarize studies on the impact of LEZs on improving air quality. This will be done separately for studies based analyses of monitoring data and studies based on analyses using air quality dispersion modelling. We have also included some few examples of studies where the effects on air quality of vehicle speed restrictions have been assessed.

## 7.1 Assessments based on monitoring data

Table 15 shows the resulting effects on measured air quality as a consequence of introducing LEZs in different cities. The table include the results as presented in this report above. Here we present the studies in the other cities as well as some other published analyses for Munich and Berlin.

**Table 15. Impact on air quality of low emission zones (LEZ) and Congestion charge systems (CCS) in different cities based on evaluating monitoring data. "na": not analyzed, "±0": No significant effect. Negative sign means decreased concentrations as a result of the LEZ/CCS.**

| **Reference** | **City** | $NO_2$ | $NO_x$ | $PM_{10}$ | **EC** | **Absor-bance** | **Particle number** |
|---|---|---|---|---|---|---|---|
| Atkinson et al (2009) | London, CCS | +[a)] | -[b)] | +[a)] | na | na | na |
| Birmili et al. (2012) | Leipzig, LEZ | na | na | ±0 | -30% | na | -30% |
| Boogard et al. (2012) | Five Dutch cities, LEZ | na | ±0 | na | ±0 | na | na |
| Cyrys et al. (2009) | Munich, LEZ | na | na | -5% to -12% | na | na | na |
| Ellison et al. (2013) | London LEZ | na | ±0 | -1% to -10% | na | na | na |
| Fensterer et al. (2014) | Munich, LEZ | na | na | -5% to -13% | na | na | na |
| Jones et al., (2012) | London LEZ and S-free diesel | na | na | na | na | na | -30% to -59% |
| Kelly et al. (2011) | London, CCS | ±0 | ±0 | na | na | na | na |
| Morfeld et al. (2013) | Munich, LEZ | na | na | ±0 | na | na | na |
| Panteliadis et al. (2014) | Amsterdam, LEZ | -4.9% | -5.9% | -5.8% | -12.9% | -7.7% | na |
| Rauterberg-Wulff and Lutz (2011) | Berlin LEZ | na | na | na | -24%[c)] -52%[d)] | | |
| Qadir et al. (2013) | Munich, LEZ | na | na | na | -50%[e)] | na | na |
| Nunes da Silva et al., 2014 | Lisbon, LEZ | (-6%)[f)] | na | (-16%)[f)] | na | na | na |
| Jensen et al. (2011) | Copenhagen LEZ | na | na | -5%[g)] | na | na | na |
| This report[h)] | Stockholm, CCS | na | -0.1% to -9.2%[i)] | -10.3% to +0.6%[i)] | na | na | na |
| This report | Augsburg, LEZ | na | na | ±0 | na | na | na |
| This report | Berlin, LEZ | na | na | -6% to -19% | na | na | na |
| This report | München, LEZ | na | na | -6% to -15% | na | na | na |

[a)] *Increase, but not possible to attribute to CCS per se.*

[b)] *Decrease, but not possible to attribute to CCS per se.*

[c)] *Decrease in 2008 compared to 2007 as reference year, for TC (total carbon)*

[d)] *Decrease in 2010 compared to 2007 as reference year, for TC (total carbon)*

[e)] *Decrease of average EC concentration in traffic factor*

[f)] *Unclear how much of this reduction is due to LEZ*

[g)] *For PM2.5*

[h)] *Based on Ducrocq and Deggendorfer (2014)*

[i)] *For kerb-side sites.*

### 7.1.1 London CCS

Atkinson et al. (2009) studied the impact of the London Congestion Charge System (CCS) on measured $NO_x$, NO, $NO_2$, $PM_{10}$, CO and O3 measured at roadside and background monitoring sites across Greater London. Temporal changes in pollution concentrations within the congestion zone were compared to changes, over the same time period, at monitors unlikely to be affected by the CCS. Similar analyses were done for CCS hours during weekends (when the CCS was not operating). They could not attribute the changes to the CCS per se since the scheme was introduced concurrently with other traffic and emissions interventions which might have had a more concentrated effect in central London.

Kelly et al. (2011) compared geometric means for the 2 years before and the 2 years after the CCS was introduced in London. Temporal changes within the CCS area were compared with changes, over the same period, at similarly sited (roadside or background) monitors in a control area 8 km distant from the center of the zone. When compared with data from outside the zone, we did not find evidence of temporal changes in roadside measurements of $NO_x$, NO, and $NO_2$, nor in urban background concentrations of $NO_x$. Although based upon fewer stations, there was evidence that background concentrations of $PM_{10}$ and CO fell within the CCS area compared with outside the zone.

### 7.1.2 London LEZ

Ellison et al. (2013) report that $PM_{10}$ concentrations have dropped by up to around 3% in the LEZ of London compared to just over 1% outside the area despite an overall growth in freight vehicles operating in London. They argue that the number of pre-Euro III vehicles has dropped and that this has been coupled with a switch from rigid vehicles to light commercial vehicles and articulated vehicles. For $NO_x$ concentrations they did not find any discernible differences.

It is also interesting to mention that introduction of “sulphur free” diesel fuel for heavy goods vehicles in the London Low Emission Zone and in Birmingham, was shown by Jones et al. (2012) to reduce particle number concentrations by 30 to 59% over a period of a few months in 2007.

### 7.1.3 Leipzig LEZ

Birmili et al. (2012) studied the effect on air quality of a Low Emission Zone in Leipzig, which was the first city in Germany where the highest regulation level (Euro 4 (or better) required for diesel cars) in a LEZ was introduced immediately after implementation of the measure. Trend analyses suggest a decrease in BC and particle number concentration (diameter 50-100 nm) in the vicinity of roads as a result of the LEZ. The decrease seemed to be first, an effect of decreasing traffic volumes in the area of the LEZ and only second, an effect of reductions in the vehicles' exhaust emission factors. For $PM_{10}$ mass concentrations no decrease could be observed.

Trend analyses presented by Rasch et al. (2013), suggest a decrease in BC and particle number concentration (diameter 50 to 100 nm) in the vicinity of roads as a result of the LEZ. The decrease was attributed to both decreasing traffic volumes in the area of the LEZ and to an effect of reductions in the vehicles' exhaust emission factors; the latter being of less importance. For $PM_{10}$ mass concentrations, no decrease could be observed. They conclude that the LEZ can be very effective in reducing the ambient concentrations of health-related particulate matter parameters BC and particle number, even if the beneficial effects are not evident from the legal parameter $PM_{10}$ mass concentration (Rasch et al., 2013).

### 7.1.4 Munich LEZ

In the first study estimating the LEZ impact in Munich, relative reduction of $PM_{10}$ mass concentration up to 12.3 % at traffic monitoring sites and about 5 % at urban background site was found (Cyrys et al., 2009). The analysis was based on the comparison of relative $PM_{10}$ concentration changes by a reference station. Such analysis of the quotient between the specific monitoring station and the reference station neglects the uncertainty of the measurements at the reference station. Furthermore, this analysis was applied on a rather short time period. Morfeld et al. (2013) analysed the same data set (it means short time period) using regression analyses for matched observations of subsequent years and have not found significant effects of the LEZ. In a study published very recently by Fensterer at al. (2014) the time period under investigation was extended and the statistical approach improved. The $PM_{10}$ concentrations before and after the LEZ implementation were compared by applying of a semiparametric statistical model with first-order autoregressive errors on data in a time resolution of one hour. The estimated $PM_{10}$ levels were adjusted for $PM_{10}$ exposure at the reference station, wind direction, season, time throughout a week, and public holidays. The results were comparable to Cyrys et al. (2009): the reduction of $PM_{10}$ concentration was larger at a traffic monitoring site (13.0 %) and smaller in urban background (4.5 %). Please note that in Munich a transit ban for heavy-duty vehicles was introduced some months before the Low Emission Zone became effective. Consequently, all analysis from Munich report the common effect of the implementation of LEZ and transit ban for trucks, and such combination is not that common.

Qadir et al. (2013) sampled $PM_{2.5}$ before (2006/2007) and after (2009/2010) the implementation of the LEZ. The samples were analyzed for carbon fraction (EC/OC) and particulate organic compounds (POC). Positive matrix factorization (PMF) was used to identify the main sources of POC. Emissions from traffic, solid fuels combustion, cooking and mixed source were separated. The contribution from traffic to $PM_{2.5}$ was decreased about 60% after the implementation of the LEZ. The average concentration of EC from traffic decreased by 50% from 1.1 to 0.5 mg/m$^3$ after the implementation of the LEZ.

### 7.1.5 Berlin LEZ

Berlin is one of the few European cities, where organic carbon (OC) and elemental carbon (EC) concentrations are measured at major roads since the1990s. Since the introduction of the LEZ local traffic in Berlin contributes less to the total carbon (TC=OC+EC) concentrations measured inside and outside the LEZ (Rauterberg-Wulff and Lutz, 2011). As a result of LEZ implementation the emission of diesel particles decreased in 2010 by 58% compared to a business as usual scenario. This leads to a veritable improvement of the air quality in Berlin as far as meteorological effects were considered as well. The evaluation of air quality data showed that in 2008, when stage 1 of the LEZ was established, TC concentrations decreased between 21 and 24 % compared to 2007. After implementation of stage 3 in 2010, TC concentrations decreased by 52 % compared to 2007. As the atmospheric conditions in the years 2008, 2009 and 2010 were more stagnant when compared to 2007; the authors attribute these results to the reduced TC emissions from traffic.

### 7.1.6 Five cities in Holland

Boogard et al. (2012) found statistically non-significant reductions in traffic related pollutants ('soot', $NO_x$, Cr, Cu and Fe) at street sites compared to control sites, following implementation of LEZs for heavy duty vehicles in five Dutch cities. Only for $PM_{2.5}$ reductions were considerably larger at urban streets (30%) and urban background locations (27%) than at the matching suburban control locations (20%). In one urban street where traffic intensity was reduced with 50%, 'soot', $NO_x$ and $NO_2$ concentrations were reduced substantially more (41, 36 and 25%) than at the corresponding suburban control location (22, 14 and 7%). In their study 'soot' was measured as reflectance on $PM_{10}$ filter samples after weighing the filters. They speculate on a number of reasons for the lack of effect on traffic related pollutants, e.g. i) the decrease in the number of old trucks was too small, ii) differences in real world emissions from trucks is smaller than anticipated in the Euro regulations; iii) counteracting or coinciding changes of concentrations due to other policies; iv) general, large decreases in emissions due to economic crisis made it hard to distinguish the LEZ effect; v) too short sampling period or that weather conditions may have blurred the LEZ effect.

Panteliadis et al. (2014) report statistically significant decreases in concentrations of $NO_2$, $NO_x$, $PM_{10}$, EC and absorbance measured at a roadside monitoring station, after the implementation of a Low Emission Zone for heavy-duty vehicles in Amsterdam. The largest effects were observed for soot particles; 12.9% reduction in EC and 7.7% reduction in absorbance.

### 7.1.7 Lisbon LEZ

Nunes da Silva et al. (2014) report 6% lower $NO_2$ and 16% lower $PM_{10}$ concentrations after the second phase of the implementation of a LEZ in Lisbon. Exactly how much of this reduction is really due to the LEZ is not assessed by the authors. It is stated that the main effect of the LEZ was to speed up vehicle fleet renewal. A survey indicated a decrease of pre-Euro and Euro 1 vehicles and increased Euro 4 and Euro 5 vehicles. No change in total traffic intensity was observed. Ferreira et al. (2012) estimated the impact of the LEZ on vehicle emissions of $PM_{10}$ (not including non-exhaust emissions) and $NO_x$. For their most aggressive scenario they estimated that (exhaust) $PM_{10}$ may be reduced by 25% to 34% and $NO_x$ by 1% to 7%.

### 7.1.8 Copenhagen LEZ

Jensen et al. (2011) assessed the effect of a LEZ in Copenhagen on concentrations of a number of pollutants (EC, PAH, $PM_{2.5}$, particle numer concentration, $NO_x$, VOC's and CO) using a regression analyses for a period of three years. For $PM_{2.5}$ the Low Emission Zone is estimated to have reduced concentrations at H.C. Andersens Boulevard by about 0.7 µg/m$^3$ during the period January 2, 2008 to December 15, 2010. For comparison, the street concentration is about 14.3 µg/m$^3$ and the street contribution is about 5.8 µg/m$^3$ in the beginning of 2008.

### 7.1.9 Milan LEZ

Invernizzi et al. (2011) compared the impact on BC with that on PM mass concentrations of a "Ecopass" zone (charge for pre-Euro 4 vehicles) and a central pedestrian zone in Milan, Italy. They showed that the effect of the regulation had highly significant effects on BC, while $PM_{10}$, $PM_{2.5}$ and $PM_1$ did not show significant differences in the different zones.

## 7.2 Modelling studies

There are several assessments based on modelling changes in air quality upon implementation of LEZs or other traffic restrictions. Above we presented results for Stockholm based an earlier study by Johansson et al. (2009) and new model calculations for Augsburg. Table 16 summarizes results from other modelling studies.

### 7.2.1 London

Beevers and Carslaw (2005) estimated reductions in emissions of the Congestion Charge System in London. They used detailed traffic data and an emission model and found that total $NO_x$ emissions fell by 12% in the charging zone between 2002 and 2003, but at the same time it increased on the ring road outside the zone by 1.5%. $PM_{10}$ emissions fell by 11.9% in the zone and fell by 1.4% on the ring road. The reductions are due to less traffic but also as a result of increased vehicle speeds. They did not assess the effects on the concentrations. Increased bus kilometers and introduction of particle traps have partly offset the decreased emissions from cars.

Carslaw and Beevers (2002) modelled impacts on $NO_x$ and $NO_2$ of different LEZ scenarios in central London; i) removing all pre-Euro I vehicles (increasing Euro II and III to conserve total number of vehicles), ii) removing pre-Euro I and LGVs and pre-Euro III HGVs and buses (increasing Euro II and III vehicles to conserve total number of vehicles and LGVs and assuming all HGVs and buses are Euro III). They also estimated effects on NO and $NO_2$ of decreasing road traffic by 10% and 20% in central London. Scenario (i), removing all pre-Euro I reduced $NO_x$ concentrations by 3-7% and $NO_2$ by 1-3%. Scenario (ii), removing pre-Euro I and LGVs and pre-Euro III HGVs and buses, reduced $NO_x$ by 11-21% and $NO_2$ by 3.6-11%. In these scenario calculations it was assumed that changes in in vehicle stock only occur on the selected roads in central London, and not changing the background concentrations. Their scenarios were made for 2005, and when they compare with predicted concentrations in 2010 without implementing any LEZ (expected cleaner vehicle technology based on legislation), they found that the levels 2010 would be similar to those in 2005 if the most effective LEZ would be introduced.

The impact of the congestion charging zone (CCZ) introduced in the inner city of London, has been modelled in several studies. Based on model calculation reported by Tonne et al. (2008) the CCZ in London should result in a reduction of 0.2 $\mu g/m^3$ for $PM_{10}$ and 0.7 $\mu g/m^3$ for $NO_2$.

Later, Kelly et al. (2011) modelled changes in concentrations of $NO_x$, $NO_2$, and $PM_{10}$ across the congestion charge zone in Greater London under different traffic and emission scenarios for periods before and after introduction of CCS. Comparing model results within and outside the zone suggested that introducing the CCS would be associated with a net 0.8-µg/m3 decrease in the mean concentration of $PM_{10}$ and a net 1.7-ppb decrease in the mean concentration of $NO_x$ within the zone. In contrast, a net 0.3-ppb increase in the mean concentration of $NO_2$ was predicted within the zone; this was partly explained by an expected increase in primary $NO_2$ emissions due to the introduction of particle traps on diesel buses (one part of the improvements in public transport associated with the CCS).

**Table 16. Impact on air quality of low emission zones (LEZ) and congestion charge systems (CCS) in different cities based on dispersion modelling. "NA": Not Analyzed, "±0": No Effect. Negative sign means decreased concentrations as a result of the LEZ/CCS.**

| Reference | City(ies) | $NO_2$ | $NO_x$ | $PM_{10}$ | EC |
|---|---|---|---|---|---|
| Carslaw and Beevers [a] (2002) | London, LEZ | LEZ1: -1% to -3% LEZ2: -3.6% to -11% | LEZ1: -3% to -7% LEZ2: -11% to -21% | na | na |
| Cesaroni et al., 2012 | Rome, Traffic restrictions+LEZ | Up to -23% | na | Up to -10% | na |
| This report[b] | Stockholm, CCS | +1% to -7% | +2% to -12% | +1% to -7% | na |
| Burman and Johansson, 2001 | Stockholm, LEZ | -0.5% to -1.5% | -0.7% to -2.5% | 0% to -1.6%[f] | na |
| Kelly et al. (2011) | London, CCS | +0.3 ppbv | -1.7 ppbv | -0.8 $\mu g/m^3$ | |
| Keuken et al. (2012) | Amsterdam, LEZ | na | na | na | -0.025 $\mu g/m^3$ [c] |
| Keuken et al. (2012) | Rotterdam, Speed restrictions | na | na | na | -0.05 to -0.2 $\mu g/m^3$ [d] |
| Jensen et al. (2011) | Several streets in 4 cities in Denmark, LEZ | -3% to -11% | na | -0.2 $\mu g/m^3$ [e] (1%) | na |
| Tonne et al. (2008) | London, CCS | -0.7 $\mu g/m^3$ | na | -0.2 $\mu g/m^3$ | na |
| This report | Augsburg, LEZ | ±0 | ±0 | ±0 | na |

[a] *Two different LEZ scenarios, see text.*

[b] *Based on Johansson et al. (2009)*

[c] *Depending on distance from road*

[d] *Population avergare exposure*

[e] *Average for all cities and all streets included in the calculations for 2010*

[f] *For $PM_{2.5}$.*

### 7.2.2 Rome

Cesaroni et al (2012) calculated effects on concentrations of $PM_{10}$ and $NO_2$ at street level as a result of a ban of vehicles in the historical center of Rome (limited traffic zone, LTZ) and restricted vehicle entries in a larger area outside the LTZ (ban of old diesel and gasoline vehicles). Model calculations showed that concentrations were reduced by 10% and 23% for $PM_{10}$ and $NO_2$ respectively, in the most affected areas.

### 7.2.3 Rotterdam and Amsterdam

Keuken et al. (2012) modelled impact of speed management in Rotterdam and concluded it to be particularly effective at reducing EC emissions on working days on a motorway. The higher the proportion of heavy duty vehicles and the higher the ratio of free-flowing to congested traffic after speed management compared to before, the more effective speed management is at improving the air quality near a motorway. They also concluded from a study in Amsterdam that a low emission zone for heavy duty vehicles made an almost negligible contribution to reducing EC emission. The main reason was the low proportion of highly polluting heavy duty vehicles, which contribute less than 5% of the total EC emission from traffic. They state that the overall traffic volume is likely to be the only effective measure available at present for reducing the health impact on the population living along inner-urban roads with intense traffic. Keuken et al. (2012) also estimated the health benefits of speed management and LEZs (see below).

### 7.2.4 Denmark

Jensen et al. (2011) made very extensive calculations of the effect on air quality of LEZs in several cities Denmark for 2010, 2015 and 2020. Calculations were performed for 138 busy streets in Copenhagen and Frederiksberg, 55 streets in Aarhus, 40 streets in Odense, and 31 streets in Aalborg. They showed that the LEZs reduced a number of pollutants, to varying degrees depending on the heavy duty traffic share and traffic emission contribution to the total concentrations. Calculated $NO_2$ reductions for 2010 were 4% for HC Andersen Boulevard and 3% for Jagtvej in Copenhagen, 11% for Banegårdsvej in Aarhus, 4% for Albanigade in Odense, and 7% for Vesterbro in Aalborg. Differences are primarily due to different shares of heavy-duty vehicles. The average effect on $PM_{2.5}$ and $PM_{10}$ were 0.2 µg/m$^3$ for all cities and all streets and the maximum effect was 0.7 µg/m$^3$.

Also they showed that the number of exceedances of the limit values for $NO_2$ were reduced. In Copenhagen (138 streets) the zone reduced the number of exceedances from 47 to 29 in 2010. In Aarhus the reduction was from 20 to 11 for the 55 streets included in the calculations.

### 7.2.5 Stockholm, LEZ

Burman and Johansson (2001) estimated the effect of the low emission zone in Stockholm. The zone was implemented 1996 and effects were calculated for year 2000 based on air quality dispersion modelling for two scenarios; i) compliance according to manually registered vehicles in the zone, ii) full compliance. Table 16 shows calculated percentage reductions of up to 1.5%, 2.5% and 1.6% for $NO_2$, $NO_x$ and $PM_{2.5}$, respectively, if there would have been 100% compliance. Burman and Johansson (2001) also estimated the effect on exhaust-PM to be -4.0% (-0.5% to -12%).

## 7.3 Health benefits of LEZs

### 7.3.1 Estimates of potential health benefits by modeling

There are much fewer studies that have estimated the health benefits of LEZs. Tonne et al. (2008) used a combination of dispersion modelling and regression calculations to analyse the air pollution and mortality benefits of the London congestion charge scheme (CCS). They concluded that the CCS lead to reductions in concentrations, although modest across Greater London, but greater in the charging zone wards. Predicted health benefits in the charging zone wards were 183 years of life per 100 000 people assuming conditions would persist over 10 years.

As mentioned previously Johansson et al. (2009) modelled impact of the congestion charge system in Stockholm, and they also assessed the health benefits by estimating the effects on mortality using a relative risk factor for $NO_x$ (Johansson et al., 2009). The total population exposure in Greater Stockholm area (35 x 35 km with 1.44 million people), was estimated to decrease with a rather modest 0.23 μg/m$^3$. Based on a long term epidemiological study, that found an increased mortality risk of 8% per 10 μg/m$^3$ $NO_x$, it was estimated that, 27 premature deaths would be avoided every year. According to life-table analysis this would correspond to 206 years of life gained over 10 years following the trial if the effects on exposures would persist. The effect on mortality was attributed to road traffic emissions (likely vehicle exhaust particles); $NO_x$ was regarded as an indicator of traffic exposure. They point out that mortality effects is only “the tip of the ice-berg” since reductions are expected in both respiratory and cardiovascular morbidity.

Keuken et al, (2012) estimated life years gained as a result of a traffic measure. They modelled reductions in EC concentrations after implementation of a speed management zone on a motorway in the city of Rotterdam. Those living within 400 m of the motorway gained up to 3 months of life expectancy, depending on their distance from the motorway. The health impact assessment was performed, assuming a decrease in life expectancy of 3.5 months due to lifetime exposure to each 500 ng EC per m$^3$. They also calculated EC concentration reductions as a result of the low emission zone in Amsterdam and estimated a population-weighted, average gain of 0.2 months in life expectancy. This was compared to the potential gain of 2.9 months due to exposure to all traffic emissions of EC in Amsterdam. It is concluded that on motorways speed management is an effective measure, while a Low Emission Zone as implemented in Amsterdam, is less effective to reduce health effects of road traffic emissions.

### 7.3.2 Monitored health benefits

There seem to be very few cohort or epidemiological studies available that can be used to assess health benefits of LEZs. The EXHALE study (Exploration of Health and Lungs in the Environment) run by Kings College in London involves health assessments in 8-9 year-old children at selected schools where one of five has a diagnosis of asthma. The assessments include measurements of respiratory health, biomarkers of exposure to traffic-related air pollution, genetic susceptibility to the effects of air pollution, and systemic response to air pollution. The health data will be linked with modelled air quality data, provided by ERG's modelling team, to provide a comprehensive picture of the effect of traffic-related air pollution on children's health, and the impact of the LEZ on this. To our knowledge no results have been published yet.

Kelly et al. (2011) analyzed the oxidative potential of particles on filters collected at several road side and urban background sites across London. The inhalation of PM with more oxidative potential could increase the risk for chronic respiratory and cardiovascular diseases in the population. It was concluded that the oxidative potential varied markedly and was higher at road side locations compared to urban background. Higher concentrations of copper, barium and bioavailable iron could have contributed to the higher oxidative potential at road side locations.

## 7.4 Other beneficial effects

Other potential benefits of LEZ and CCS are lower emissions of i) pollutants associated with effects on the environment like acidification/eutrophication, ii) precursors of oxidant formation like ozone (which may cause damage on crops and other vegetation), and iii) pollutants that have climate impacts. LEZ may also speed up the change to quieter vehicles potentially reducing noise pollution in cities. There seem to be very few studies that have actually made quantitative assessments of such effects.

## 7.5 Implementation and effectiveness of LEZ/CCS

The effectiveness of a LEZ for reducing air pollution depends upon a number of factors including what emission standards are implemented, current vehicle composition, the size of the area and to what degree the regulations are being fulfilled. The latter depends on e.g. communication of the rules to the drivers, the legal framework, vehicle exemptions and how the enforcement is made. There are no uniform regulations, but several countries have national regulations (Sadler Consultants, 2011; Ecorys, 2014). Most LEZs regulate emissions from heavy-duty vehicles (HDVs) and allow vehicles to meet the set standards by the retrofitting of diesel particulate filters (DPFs) (Sadler Consultants, 2011). Enforcement is usually through manual inspections by police or traffic wardens, but in some cases cameras with charging using automatic number plate recognition (APNR). Compliance of LEZ regulations may be higher if APNR is used (Ecorys, 2014). Some LEZs are based on charging. In London, non-compliant vehicles can pay to enter the zone, and in Milan access to the zone is based on a charge depending on the time of the day (Ecorys, 2014).

The effectiveness of a LEZ also depends on the time perspective. In some years after the LEZ has been introduced its effect will decrease due to the ongoing turn-over of the vehicle fleet with newer technologies having lower emissions according to the legislation. As shown by Carslaw and Beevers (2002) implementation of their most ambitious LEZ in central London 2005, would give the same improvements in air quality 2005, as predicted in 2010 without doing anything. A key question posed by the authors, is if the reductions achieved five years earlier are significant enough to warrant the high costs of implementing them.

The emissions of pollutants which are of non-local origin are not affected at all, and non-exhaust sources will be much less affected than pollutants that are mainly coming from local exhaust. Non-exhaust pollutants may be affected if traffic intensity or driving conditions change. For example, less congestion may reduce brake wear emissions, but may also increase mean vehicle speed which in turn affect suspension of road dust. This means that the efficiency of a LEZ depends on what pollutant that is studied. Elemental carbon (respectively Black Carbon BC) or $NO_x$ which are emitted mainly by traffic are more influenced by local road traffic exhaust and thus more affected than $PM_{10}$ or $PM_{2.5}$ to detect and demonstrate the efficacy of traffic restrictions.

Lefebvre et al. (2011) showed that the effect of speed reductions during “smog episodes” on elemental carbon (EC) is more important than if $PM_{10}$ or $PM_{2.5}$ is considered. So from the viewpoint of health benefit such actions may be well justified. They showed that in the very dense highway network in Flanders with a 60% share of diesel cars (the highest in Europe) a speed limit reduction from 120 to 90 km/h during winter smog episodes on selected sections of Flemish highways leads to a significant decrease of the EC concentrations near those highways.

Nitrogen dioxide, $NO_2$, may not be a suitable indicator to evaluate the effect of traffic abatement measures as some actions, such as filters on diesel fuelled vehicles, may increase $NO_2$ levels (Millstein and Harley 2010), but at the same time such actions may be beneficial for health as they decrease exposure to combustion PM from traffic.

Many studies have pointed out the fact that it can be difficult to detect improvements in air quality of LEZs due to e.g. other changes occurring at the same time, variable meteorological conditions obscuring the effects of the LEZ under study. In most cases $PM_{10}$ and/or $NO_2$ has been used as indicators, which may not have been optimal as discussed above.

Another issue that is difficult to consider is the potential effect of road traffic displacement, i.e. non-LEZ-compliant vehicles avoiding the LEZ and possibly even increasing their kilometers driven, and thereby somewhat compensating for the lower emissions inside the zone. This complicates assessments of the efficacy of LEZs based comparisons of concentrations measured inside and outside the zone.

The study by Kelly et al. (2011) of the London Congestion Charging System CCS illustrate the risk that introduction of vehicle regulations may even have detrimental effects on air quality – the shift from gasoline- to diesel-powered vehicles and emission control on buses that altogether led to an increase in $NO_2$ concentrations. For $NO_2$, the non-linear photochemical reactions result makes analysis of the impact of vehicle emissions on ambient concentrations very complicated (Carslaw and Beevers, 2002). For $PM_{10}$ and $PM_{2.5}$, non-exhaust emissions may be a substantial part of locally generated traffic emissions but not regulated by LEZs. A combination of traffic and exhaust emission restrictions may be the most advantageous, since that reduces both exhaust and non-exhaust emissions.

There are also many uncertainties in assessments and prognoses of the effect of a LEZ since estimates based on modelling on emission factors for later Euro standards that have not been tested and may have higher emissions in real world driving compared to in the test programs.

# 8 Some final recommendations

LEZ and CCS and other traffic reduction measures have potentially large health benefits. More and more scientific studies have been using monitoring data to assess the potential air quality benefits. Such assessments has been difficult due to the many factors that affect air pollution levels, In particular how to compensate for variable meteorology, uncertainties in measurement data and how to consider effects of changes in traffic emissions solely by the LEZ and not by other causes.

In order to make a proper evaluation of LEZ or CCS on air quality it is very important to monitor the impacts on traffic intensities and vehicle composition, not only inside the zone, but also outside. Traffic intensities are often measured in cities at some points, but may need to be complemented. Vehicle composition changes may be monitored manually or by automatic number plate recognition (ANPR) systems. Such systems are also effective for enforcement of the LEZ regulations. APNR is likely to be more efficient than manual enforcement methods. They may be more expensive in the short term but may be less expensive once in operation.

Ongoing fleet turnover need to be considered to maintain effectiveness of a LEZ. Reliable and effective vehicle emission standards are necessary for deciding on the regulations of a LEZ. Real world emissions are sometimes larger than expected according to the emission standards making prognoses of LEZ impacts uncertain.

Air quality monitoring in cities usually include $PM_{10}$ and $NO_2$, as these pollutants are regulated in the EU directive. Exceedances of the limit values for these pollutants are also often the main driver for introducing a LEZ. But the benefits of LEZ for the health of the population may be significant even if the reductions in $PM_{10}$ and $NO_2$ concentrations are small, since LEZ reduce exhaust combustion particulate (including black carbon) that may have the highest adverse health impacts. It is therefore recommended to complement measurement programs with pollutants that are better specific markers for vehicle exhaust emissions. In this respect soot particle concentrations (measured as BC, EC or absorbance) or particle number concentrations are better indicators than $PM_{10}$ and $NO_2$. In densely trafficked areas soot particles are mainly from diesel exhaust emissions, whereas $PM_{10}$ is due to many different sources including non-exhaust emissions. Thus exhaust particles is only a small fraction of $PM_{10}$. $NO_x$ is a better marker than $NO_2$, as $NO_2$ is influenced by photochemical reactions and only a small fraction is due to exhaust emissions. But since retrofitting of diesel vehicles may even increase $NO_2$ emissions it is important to assess LEZs impacts on $NO_2$ as well.

There are several other measures than LEZ and CCS that have been shown to significantly improve air quality. LEZ affect only exhaust components, but CCS and other traffic access restrictions aim at reduce traffic intensities and may affect all traffic emitted pollutants. In this report we have included some studies were speed management have been implemented, but there are many more examples of this not reported here (e.g. Keller et al., 2008; Gonçalves et al., 2008; Dijkema et al., 2008; Zhang et al., 2011).

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