Chapitre 5

5 Modélisation de l'impact du trafic sur la qualité des eaux de ruissellement urbaines

Ce chapitre est consacré à l'élaboration des données d'entrée nécessaires au modèle de qualité de l'eau et à la simulation de la propagation des polluants des eaux de ruissellement en milieu urbain. Les données essentielles sont les données météorologiques et les caractéristiques du bassin versant (en particulier l'occupation du sol) pour la simulation quantitative de l'eau, l'accumulation et le lessivage superficiel des polluants en vue de la modélisation de la qualité de l'eau. Dans ce chapitre, on présente trois méthodes différentes pour modéliser l'impact du trafic sur la qualité des eaux de ruissellement.

La première méthode utilise des données expérimentales de dépôts atmosphériques liés au trafic à proximité d'une route. Cette partie a été publiée dans des actes d'un congrès international : Fallah Shorshani M., Bonhomme C., Petrucci G., André M., Seigneur C. (2012) Road traffic impact on water quality in an urban catchment (Grigny, France), *9th International Joint IWA/IAHR Conference on Urban Drainage Modelling*, Belgrade, Serbie (Annexe C).

La deuxième méthode est basée sur l'utilisation d'un modèle « boîte » qui calcule les concentrations de polluants et les dépôts atmosphériques. Cette partie a été publiée dans une revue internationale avec comité de lecture : Petrucci G., Gromaire M. C., Fallah Shorshani M., Chebbo G. (2014). Non-point source pollution of urban stormwater runoff: a methodology for primary sources' analysis, *Environmental Science and Pollution Research*. doi:10.1007/s11356-014-2845-4 (available online) (Annexe D).

La troisième méthode crée une chaîne de modélisation complète qui inclut les modèles de trafic, d'émissions, de qualité de l'air et de qualité de l'eau. Cette chaîne de modélisation est évaluée avec des données expérimentales du bassin versant de Grigny en banlieue parisienne. Cette partie a été publiée dans une revue internationale avec comité de lecture : Fallah Shorshani M., Bonhomme C., Petrucci G., André M., Seigneur C. (2014). Road traffic impact on urban water quality: a step toward integrated traffic, air and stormwater modeling, *Environmental Science and Pollution Research*, Volume 21, Numéro 8, Pages 5297-5310.

5.1 Couplage basé sur les données expérimentales de dépôts atmosphériques

Cette étude estime l'effet de l'impact du trafic sur les eaux de ruissellement du bassin versant de Grigny en région parisienne, pendant les années 2009 et 2010. On ne réalise pas ici une chaîne de modélisation complète, car nous utilisons directement des données expérimentales de flux de dépôts de polluants atmosphérique (Promeyrat, 2001) au lieu de simuler ces dépôts atmosphériques. L'étude se concentre sur certaines contaminations des eaux de ruissellement induites par le trafic comme celles liées au cadmium (Cd), plomb (Pb) et zinc (Zn). Le bassin versant de Grigny est divisé en 20 sous-bassins versants. Chaque sous-bassin versant est supposé être constitué de quatre catégories de milieux (espaces verts ou végétalisés, toitures, routes, et autres surfaces) pour tenir compte de la variabilité d'occupation des sols. La situation géographique, la division du bassin versant et la rose des vents sont présentées en Figure 5.1.

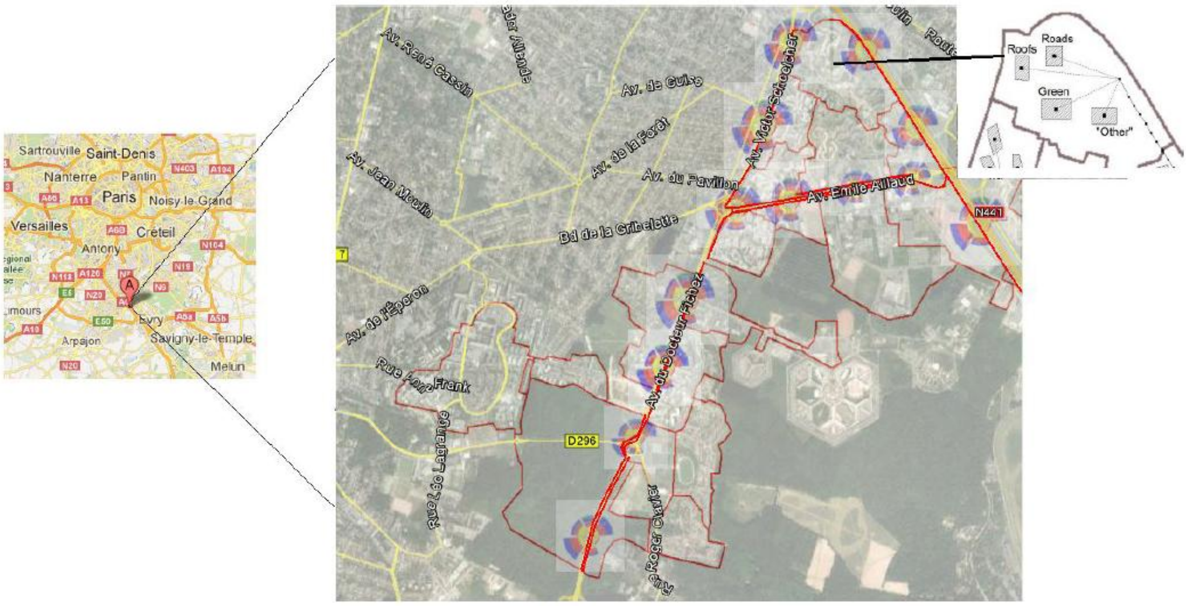

Figure 5.1. Situation géographique et détails du bassin versant de Grigny.

Le modèle utilisé pour la simulation de la quantité et de la qualité de l'eau est SWMM 5 (Rossman, 2010). La modélisation et la calibration de la quantité de l'eau ont été réalisées par Petrucci et al. (2013). Les simulations de la qualité de l'eau comprennent l'accumulation de polluants pendant les périodes sèches et leur lessivage au cours des événements de précipitation. Différentes approches mathématiques sont disponibles pour représenter les processus qui régissent l'accumulation de polluants et leur lessivage. Une approche exponentielle est utilisée ici pour l'accumulation. Dans cette équation, l'accumulation de polluants (masse par unité de surface) dépend du taux d'accumulation quotidien (masse par unité de surface et par jour), le nombre de jours secs antécédents (jours) et le coefficient d'enlèvement (1/jour) qui est disponible dans la littérature scientifique selon le mode d'occupation de sol (Liu et al., 2010). Les taux d'accumulation quotidiens sont calculés selon le flux de dépôt de chaque source de polluant en fonction du niveau de trafic et de la direction du vent. Le dépôt de polluant suit une relation linéaire avec le volume de trafic (Brett et al., 2011). Le taux d'accumulation quotidienne peut donc être calculé selon le volume de trafic et en fonction de la rose des vents. D'après des travaux précédent sur les zones d'impact du trafic routier en bordure de route (Sabin et al., 2006 ; Loubet et al., 2010), les taux d'accumulation quotidienne pour chaque sous-bassin versant ont été calculés sur des zones impactées par la circulation routière qui s'étendent jusqu'à 240 m de la route. Chaque route impacte donc une fraction du sous-bassin versant en fonction de sa localisation par rapport au sous-bassin versant et selon la direction du vent. Les autres parties du sous-bassin versant sont supposées être impactées seulement par les dépôts de fond. Pour les zones en bordure de route (< 240 m), le flux de dépôt atmosphérique est calculé en utilisant ces données expérimentales de Promeyrat (2001), qui ont été obtenues sur un autre site et qui sont donc pondérées par le rapport des débits de trafic estimés pour ces différentes sites. La charge de lessivage des polluants est proportionnelle à une fonction du débit d'eau (mm/h) qui dépend d'un coefficient d'ajustement, au coefficient de lessivage et à l'accumulation de polluants. La simulation de la qualité de l'eau peut alors être réalisée à l'aide de ces paramètres et de données d'entrée par le modèle SWMM. Une analyse de sensibilité pour tous les coefficients du modèle d'accumulation et de lessivage a aussi été effectuée.

La comparaison des deux cas avec et sans trafic local confirme un effet significatif de la circulation automobile sur la contamination de l'eau. Les concentrations maximales de Cd, Pb, et Zn à l'exutoire sont respectivement 2,12; 284,6; et 1758 µg L^{-1} (Figure 5.2.a). Ces valeurs pour le cas sans trafic (impacté seulement par les dépôts de fond) sont respectivement 0,78 µg-Cd-L^{-1}, 47,72 µg-Pb-L^{-1}, et 835,17 µg-Zn-L^{-1} (Figure 5.2.b). Les concentrations moyennes dues au trafic et à la concentration de fond sur une période de deux ans (2009-2010) sont 0,08 µg-Cd-L^{-1}, 6,33 µg-Pb-L^{-1}, et 79 µg-Zn-L^{-1}. Dans le cas où on ne prend pas explicitement en compte l'impact du trafic, ces valeurs sont 0,06 µg-Cd-L^{-1}, 4,01 µg-Pb-L^{-1}, et 70 µg-Zn-L^{-1}. Compte-tenu de l'incertitude des paramètres du modèle, ces résultats sont réalistes et comparables aux mesures effectuées par Sabin et al. (2005) à Los Angeles, qui donnent des concentrations annuelles mesurées moyennes (± écarts-types) de 160 ± 130 µg-Zn-L^{-1} et 12 ± 10 µg-Pb-L^{-1}.

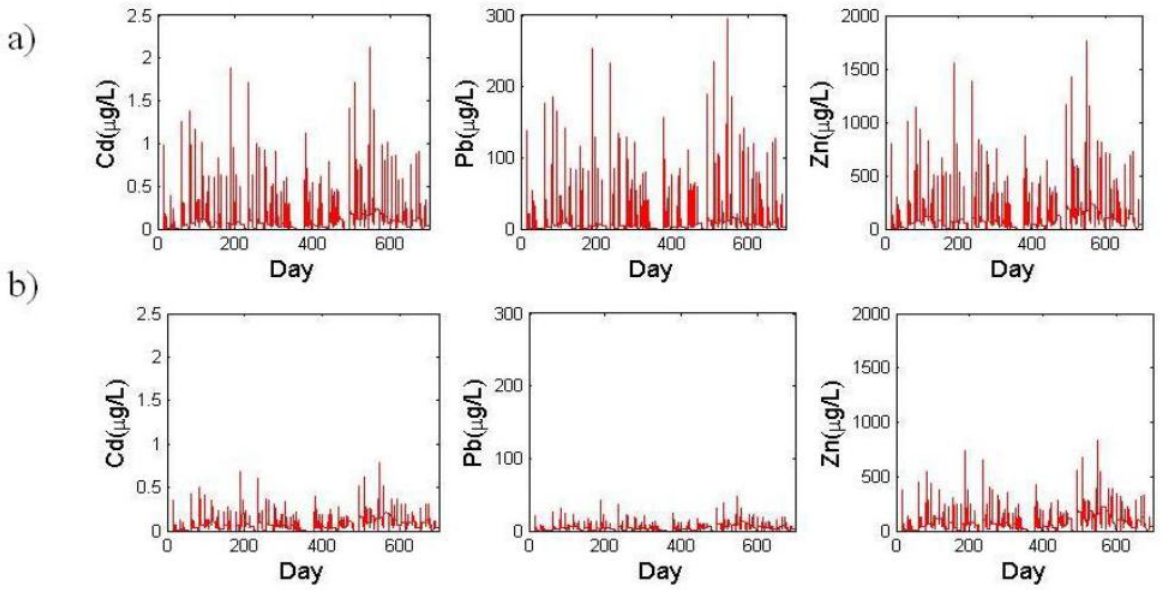

Figure 5.2. Concentrations de métaux lourds (mg / L) à l'exutoire du bassin versant de Grigny : a) avec prise en compte du trafic local et b) dues seulement aux dépôts de fond moyens en zone urbaine.

Les concentrations de zinc au moment de pics de pollution dans différents sous-bassins versants sont illustrées en Figure 5.3. Les zones les plus polluées (> 800 µg-Zn-L^{-1}) sont les sous-bassins versants imperméables près de l'autoroute très fréquentée et les sous-bassins versants influencés par les deux routes principales. Les zones les moins polluées contiennent des sous-bassins versants situés loin des routes et les grands sous-bassins versants ayant une faible fraction de la surface impactée par le trafic.

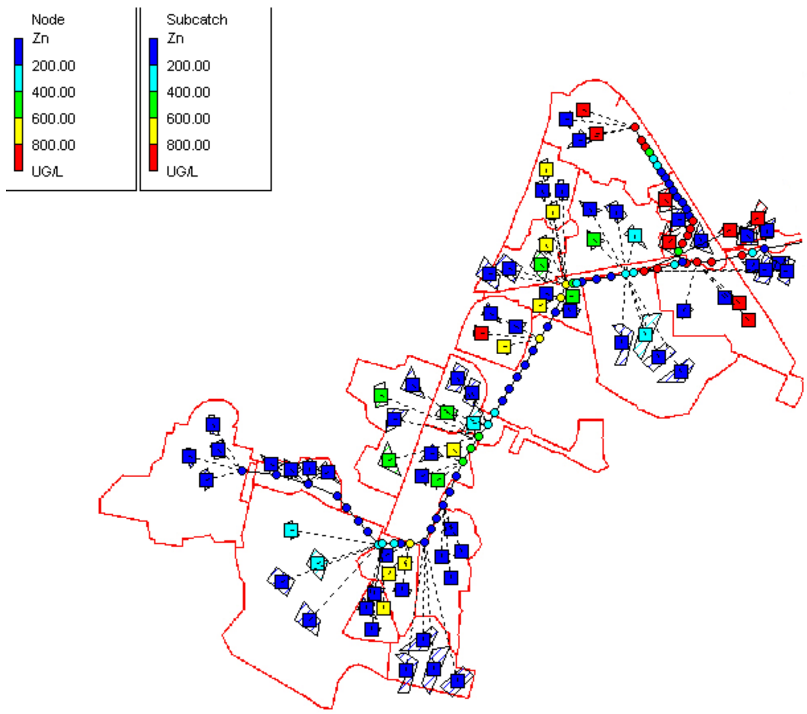

Figure 5.3. Répartition spatiale de la concentration de zinc (µg/L) issue des dépôts atmosphériques au pic de pollution, 06h30, 07/03/2010.

Ces résultats montrent que les concentrations de polluants dans les eaux de ruissellement peuvent être trois fois plus élevées pour certaines zones géographiques si on utilise une description explicite du trafic. Par conséquent, une connaissance exhaustive de la répartition spatiale des routes avec un volume important de trafic est importante pour prévoir correctement la qualité de l'eau dans les zones modélisées.

5.2 Estimation des dépôts atmosphériques à l'aide d'un modèle boîte

La méthode précédente a utilisé des données expérimentales de dépôts atmosphériques pour estimer l'accumulation des polluants sur les bassins versants. Ici, les dépôts atmosphériques sont calculés avec un modèle boîte. Les résultats sont également évalués avec des mesures effectuées sur un bassin versant en région parisienne (Sucy-en-Brie). Cette approche suppose que les concentrations atmosphériques et les dépôts sont spatialement uniformes sur le domaine de l'étude. Celui-ci est défini comme un volume parallélépipédique couvrant le bassin versant. La hauteur de la boîte s'étend en altitude jusqu'à 1000 m de la surface pendant la journée et est limitée à 50 m pendant la nuit à cause de la stabilité atmosphérique qui réduit le mélange vertical dans la couche limite atmosphérique. Ce modèle suppose également que la dynamique de l'écoulement est dominée par l'advection plutôt que par la convection (Bénarie, 1967). Les concentrations et le flux de dépôt peuvent être calculés pour chaque pas de temps (1 h) par les équations suivantes:

$$C = \frac{E}{\sqrt{LW}hu} \qquad 5.1$$

$$F_d = V_d . C \qquad 5.2$$

où C est la concentration (g/m^3), E est l'émission (g/s), L et W sont la longueur et la largeur (m) du bassin versant, h est la hauteur (m) de la couche de mélange, u est la vitesse du vent (m/s), F_d est le flux de dépôt, et V_d est la vitesse de dépôt, définie en fonction des conditions météorologiques et de la taille des particules. On suppose que la distribution en taille des PM_{10} inclut 74% de particules de diamètre inférieur à 1 µm, 10% de particules entre 1 et 2,5 µm, et 16% de particules avec un diamètre entre 2,5 et 10 µm. Les vitesses de dépôt des particules dans ces trois gammes de taille pour une vitesse de vent moyenne de 5 m/s sont respectivement de 0,2, 0,4 et 4 cm/s, selon Roustan (2005).

Les données d'entrée nécessaires au modèle sont les concentrations de fond des polluants, les données météorologiques, et les émissions provenant du trafic et du chauffage domestique. Les concentrations de fond ont été obtenues à partir de mesures effectuées dans la région parisienne (Ayrault et al., 2010; INERIS, 2000). Les valeurs des concentrations de fond sont 18,4, 6,3, 45,8, et 1,61 ng/m^3 pour Cu, Pb, Zn et HAP, respectivement. Ces concentrations de fond sont spatialement uniformes sur le domaine de l'étude et constantes dans le temps. Les données météorologiques sont basées sur des observations à la station la plus proche (aéroport d'Orly, à 10 km du bassin versant). L'utilisation de ces données est appropriée dans ce cas, en raison d'un terrain relativement plat.

Les émissions des véhicules sont estimées à partir de données de trafic et de facteurs d'émission. Les facteurs d'émission des métaux lourds sont définis d'après les résultats expérimentaux de Sternbeck et al. (2002). Pour les HAP, nous avons utilisé le modèle CopCETE (voir plus haut). Les émissions de chauffage domestique ont été estimées par Petrucci et al. (2014).

Le modèle calcule le dépôt annuel moyen par hectare. Les résultats ont été comparés avec les dépôts moyens atmosphériques mesurés sur le bassin versant, qui comprennent huit mesures effectuées entre 2011 et 2013 par Gasperi et al. (2013), ainsi qu'une estimation des dépôts dans un bassin versant voisin (Créteil) en 2001/2002 (Azimi et al., 2003, 2005). Pour les trois métaux, et en particulier Pb, les données de Créteil sont plus élevées que les mesures effectuées à Sucy-en-Brie et que les résultats de la modélisation. Ceci est probablement dû à la densité plus élevée, au trafic et à l'industrialisation du bassin versant de Créteil par rapport au bassin versant de Sucy-en-Brie, et aussi à la réduction progressive des émissions dans la dernière décennie à cause des réglementations. Par exemple, les émissions de Pb ont fortement diminué à cause de l'élimination de ce métal des essences et de ses usages actuellement plus limités dans les pièces des véhicules. Le modèle utilise les facteurs émission qui surestiment probablement les émissions actuelles de Pb et donne donc des résultats plus élevés que les valeurs mesurés sur le bassin versant. Les résultats des autres métaux sont assez satisfaisants par rapport aux mesures. En revanche, le modèle surestime les HAP. Le bon accord entre les données de Créteil et les mesures de Sucy-en-Brie suggère que les résultats de la modélisation ne sont pas fiables pour les HAP.

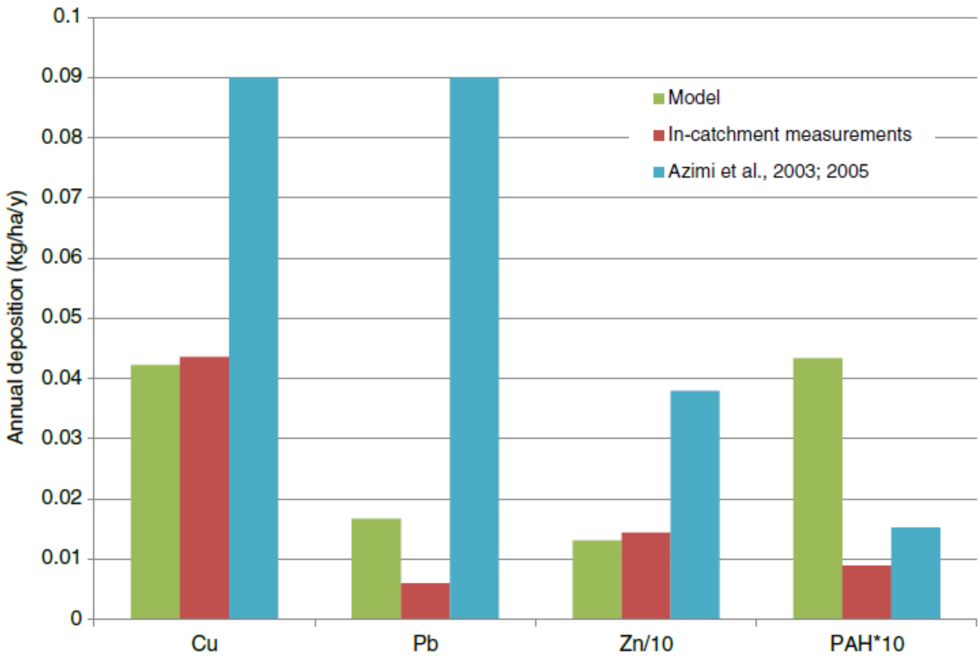

Figure 5.4. Les estimations des dépôts atmosphériques annuels sur le bassin versant de Sucy-en-Brie (Petrucci et al., 2014).

Ce modèle boîte est relativement simple et n'est donc pas capable de décrire précisément les processus de dispersion et de dépôt des métaux lourds et des HAP. Il est cependant utile pour fournir des estimations moyennes sur une période assez longue.

5.3 Chaîne de modélisation

Cette partie est consacrée au développement d'une chaîne de modélisation complète avec couplages des modèles de trafic, d'émission de polluants, de dispersion et de dépôts atmosphérique, et de contamination des eaux de ruissellement. Cette étude se concentre sur la mise en œuvre d'une chaîne de modélisation sur un bassin versant suburbain (Grigny) en région parisienne pendant la période du 2 avril au 3 mai 2012. Les polluants des eaux de ruissellement étudiés sont le cadmium (Cd), le plomb (Pb) et le zinc (Zn). Ces métaux peuvent être émis par le trafic routier.

Les volumes horaires de trafic sont déterminés à partir de comptage sur l'autoroute et une modélisation par modèle statique de VISUM pour les routes nationales. La circulation locale sur la zone résidentielle a été traitée comme une source spatialement uniforme dans le bassin versant en utilisant le nombre de véhicules immatriculés et le nombre de travailleurs qui vivent dans le domaine d'étude. Avec des données horaires de trafic, les émissions à l'échappement de ces trois métaux lourds ont été estimées par le modèle basé sur la vitesse moyenne (CopCETE). Les émissions non-échappement ont été calculées avec la méthode de COPERT4 pour l'usure des pneus et des freins. En revanche, l'usure de la chaussée n'a pas été prise en compte en raison d'un manque de données. Une grande partie de ces émissions est constituée de particules grossières qui ne restent pas longtemps dans l'air en raison de leurs sédimentations, mais celles-ci sont une source importante de contaminations des eaux de ruissellement. La contribution de l'usure des chaussées est particulièrement importante quand on s'intéresse à la contamination des eaux en HAP, mais elle est moindre pour l'étude des métaux. Par conséquent, les émissions de particules supérieures à 10 µm de diamètre aérodynamique doivent être prises en compte dans les inventaires d'émission. La dispersion atmosphérique des polluants a été calculée avec le modèle de dispersion Gaussien pour les sources linéiques de la plateforme de modélisation Polyphemus (Briant et al., 2011, 2013). Le modèle Polyphemus utilise une nouvelle approche pour réduire l'erreur de calcul lorsque la direction du vent n'est pas perpendiculaire à la route. Les concentrations des polluants dues à l'autoroute et aux routes nationales sont calculées sur des grilles rectangulaires constituées de récepteurs espacés de 50 m et spécifiques à chaque sous-bassin versant. La contribution de la circulation sur les rues résidentielles a été simulée en utilisant une approche de modèle boîte (voir ci-dessus), en supposant une dispersion atmosphérique uniforme des polluants sur le bassin versant. Le dépôt sec à chaque point récepteur a été estimé en fonction de la concentration du polluant et de sa vitesse de dépôt. En revanche, le dépôt humide est considéré spatialement homogène sur le bassin versant.

La simulation de la qualité de l'eau nécessite une simulation de la quantité d'eau parce que les concentrations de polluants dépendent du flux d'écoulement simulé. Le modèle utilisé pour effectuer l'analyse des eaux de ruissellement est SWMM 5 (Rossman 2010). La modélisation de la quantité d'eau pour le bassin versant de Grigny est réalisée en utilisant des données topographiques et des données sur l’occupation des sols. Elle est complétée par une calibration avec un algorithme génétique (Petrucci et al., 2013). Les simulations de qualité de l'eau comprennent l'accumulation de polluants pendant les périodes sèches et leur lessivage lors des événements pluvieux. Dans cette étude, l'accumulation est obtenue à partir de dépôts atmosphériques calculés avec les modèles de dispersion et de dépôts atmosphériques. Le lessivage peut ensuite être déterminé en fonction de l’accumulation des polluants, de l’intensité de la pluie et de paramétrisations (voir plus haut) qui dépendent de l’occupation des sols.

Dans ce cadre, trois simulations ont été réalisées : (1) une simulation du modèle Gaussien, avec toutes les sources d'émissions (trafic local et concentrations de fond), (2) une simulation sans aucune émission du trafic local, avec seulement les concentrations de fond, (3) une simulation du modèle boîte (voir partie précédente) avec toutes les émissions du trafic local et les concentrations de fond.

La comparaison entre la simulation avec le modèle Gaussien et les mesures expérimentales montrent que pour les taux moyens de débits massiques la valeur simulée représente 77% de la valeur mesurée pour Zn, 100% pour Cd et 211% pour Pb, ce qui démontre un impact significatif des polluants atmosphériques sur la qualité de l’eau. Le Zn provient de la corrosion des matériaux de construction, ce qui peut expliquer le fait que les concentrations simulée sont plus faibles que les concentrations mesurées car cette source n’est pas prise en compte dans le modèle. On a utilisé les facteurs émission de COPERT référencés entre 1983 et 2005 mais, actuellement, l'essence ne comporte plus de Pb en Europe et les émissions de Pb « non-échappement » ont aussi été réduites. Les résultats simulés de Pb surestiment donc fortement les mesures. Cependant, le Pb reste encore une source importante de contamination de l'eau (Gromaire et al. 2011).

La comparaison des simulations 1 (avec trafic local) et 2 (sans trafic local) montre que l’effet maximum de la concentration de fond de polluants est inférieur à 10%. La contribution du trafic local est donc dominante parce que (1) les émissions du trafic sont à proximité du sol et, par conséquent, disponibles pour le dépôt sec et (2) la présence de particules d'un diamètre supérieur à 10 µm de l'usure des équipements du véhicule (c'est-à-dire des particules avec de grandes vitesses de dépôt) produit des dépôts secs importants à proximité des routes.

Le rapport des résultats de la simulation 3 (émissions moyennées spatialement) sur ceux de la simulation 1 (émissions du trafic distribuées spatialement) montre des écarts des taux moyens des débits massiques calculés par les modèles en moyenne pour tous les événements de pluie de 24% pour Zn, 41% pour Cd et 34% pour Pb. Cela signifie qu’un modèle boîte réduit l'impact des sources de trafic local sur les concentrations des contaminations des eaux à l'exutoire du bassin versant de manière significative à cause d’une dilution exagérée des émissions du trafic dans la couche de mélange atmosphérique. Par conséquent, les différences entre les concentrations de métaux émis par le trafic obtenues avec ces deux simulations montrent qu'il est essentiel d'avoir une représentation spatiale correcte des sources d'émissions locales.

Abstract

Methods for simulating air pollution due to road traffic and the associated effects on stormwater runoff quality in an urban environment are examined with particular emphasis on the integration of the various simulation models into a consistent modelling chain. To that end, the models for traffic, pollutant emissions, atmospheric dispersion and deposition, and stormwater contamination are reviewed. The present study focuses on the implementation of a modelling chain for an actual urban case study, which is the contamination of water runoff by cadmium (Cd), lead (Pb), and zinc (Zn) in the Grigny urban catchment near Paris, France. First, traffic emissions are calculated with traffic inputs using the COPERT4 methodology. Next, the atmospheric dispersion of pollutants is simulated with the Polyphemus line source model and pollutant deposition fluxes in different subcatchment areas are calculated. Finally, the SWMM water quantity and quality model is used to estimate the concentrations of pollutants in stormwater runoff. The simulation results are compared to mass flow rates and concentrations of Cd, Pb, and Zn measured at the catchment outlet. The contribution of local traffic to stormwater contamination is estimated to be significant for Pb and, to a lesser extent, for Zn and Cd; however, Pb is most likely overestimated due to outdated emissions factors. The results demonstrate the importance of treating distributed traffic emissions from major roadways explicitly since the impact of these sources on concentrations in the catchment outlet is underestimated when those traffic emissions are spatially averaged over the catchment area.

5.3.1 Introduction

Traffic is a major source of pollution in cities and near highways. Therefore, it is essential to assess the impact of road traffic on air and stormwater pollution. To that end, numerical modelling is needed to estimate the contribution of on-road vehicles to pollutant concentrations in air and water and to provide the basis for the design of efficient emission reduction strategies. Models have been developed to address the various components of this environmental system: traffic, emissions, atmospheric pollution, and stormwater pollution. Current traffic models can predict the position and kinematic parameters of the vehicles with various levels of detail. Emission models can estimate the amount of different pollutants emitted by vehicles, albeit with some uncertainty (Smit et al., 2010). The dispersion of pollutants in the atmosphere can be simulated using atmospheric dispersion models available for application at a variety of spatial scales and with different levels of detail (Holmes et al. 2006; Zannetti 1990; Sportisse 2009). A fraction of the air pollutants deposits to surfaces by dry and wet processes. These pollutants may then be entrained by the water runoff during rainfall events, which can be simulated by hydrologic models. If models have been developed, applied and, to some extent, evaluated for each of those components, the integration of all those components to simulate the impact of road traffic on air and water pollution has been very limited to date.

Some modelling systems linking traffic flow, emissions of air pollutants and air quality modelling tools have been developed (e.g., Lim et al. 2005; Schmidt and Schäfer 1998; Hatzopoulou 2010). Although a large amount of work has been conducted to link atmospheric pollution to surface water contamination at regional and global scales for environmental issues such as acid deposition, mercury contamination and nutrients inputs causing eutrophization of water bodies, little attention has been paid to linking air and water pollution in urban areas. Since similar pollutants (e.g., polycyclic aromatic hydrocarbons (PAHs), metals) are regulated for air and water quality, it seems essential to treat air and water contamination jointly when addressing the environmental impacts of major sources of those pollutants. For example, it has been estimated that 57-100% of the trace metal load on urban surfaces is impacted by atmospheric deposition in the Los Angeles basin (Sabin et al., 2005). Traffic exhaust emissions are a major source of atmospheric pollution in urban areas, and in addition, tyre, brake, and road wear lead to the release of particulate pollutants, which are transported by stormwater runoff (Ken, 2013). To date, the link between traffic emissions, atmospheric deposition and the contamination of water runoff in urban areas has not been treated yet in a comprehensive manner.

In this work, we address a new area in the field of urban modelling by developing a new modelling method that integrates different models simulating traffic, emissions, air pollution and stormwater pollution in order to predict the level of contamination of surface waters at the scale of an urban district. The existing methods for modelling each component are presented first and the levels of detail needed for specific applications are briefly discussed. Interfaces are developed here for pollutant emissions due to on-road traffic, mass transfer between the atmosphere and surfaces, and mass transfer of deposited pollutants to stormwater runoff. This method is evaluated for the Grigny catchment near Paris, France, by a comparison of model simulations with measurements of metal concentrations at the catchment outlet. The contribution of local traffic emissions to those metal concentrations is estimated and compared to background deposition. Furthermore, the sensitivity of the modelling results to the treatment of traffic in the modelling chain (spatially-distributed versus spatially-averaged) is investigated.

5.3.2 Overview of models

The different types of models typically used are summarized below for each component (traffic, emissions, air quality and water quality).

5.3.2.1 Traffic models

Three major classes of traffic models can be identified in order of increasing complexity:

1. Static models rely on population data and predict average traffic volumes in different areas of a road network. They give vehicle fluxes on each link of the road network. These models are highly simplified, but are useful to describe the road traffic (flow and speed) over large spatial scales, such as an entire metropolitan area.

2. Aggregated dynamic models provide an explicit representation of traffic congestion by describing the temporal evolution of traffic states over a simplified network (via spatial aggregation). A typical application is the evaluation of the level of congestion, which can be used to manage traffic flow.

3. Dynamic models describe the temporal variations of traffic conditions in order to determine the positions and kinematic parameters of vehicles in a road network. Dynamic models can be classified into two main categories according to different aspects of traffic

flow operations. The macroscopic models use an aggregate representation of vehicles and continuous traffic flow; they are characterized by variables such as traffic flow and vehicle density. The microscopic models consider the time-space behaviour of individual drivers in relationship with the presence of vehicles in their proximity.

5.3.2.2 Emission models

The usual approaches for estimating the emissions associated with road traffic can be classified according to the input data, the scale of the study, and the type of pollutants being considered. These approaches can be distinguished in order of increasing complexity as follows:

1. Models relying on fuel sale data
2. Models relying on annual average traffic volumes per vehicle categories
3. Models relying on average speed of traffic
4. Models relying on traffic situations
5. Models relying on traffic-related variables
6. Models providing emissions from various driving cycle variables
7. Models relying on speed chronology, also known as instantaneous emission models

Models of categories 1 and 2 can be used for large-scale emission inventories (e.g., national level inventories). Models of categories 3 and 4 provide more accurate information and cover the major emission processes and most pollutants from a single road up to an entire city. Models of category 5 require traffic flow variables for each road and category 6 is defined by individual vehicle movement data. These two categories of models are restricted to specific conditions and a limited number of pollutants. Models of category 7 represent explicitly the vehicle emission behaviour by relating emission rates to vehicle operation (engine power, speed, and acceleration) during a series of short time steps. They only address a limited number of pollutants (typically regulated gaseous pollutants) from vehicle exhaust.

5.3.2.3 Air pollution models

Air pollution models (also referred to as air quality models or atmospheric chemical-transport models) calculate atmospheric pollutant concentrations from inputs (emissions, meteorology, terrain, initial and boundary conditions) and mathematical representations of the physico-chemical processes governing the temporal evolution and spatial distribution of the pollutant concentrations. The different approaches available are the following (Zannetti, 1990; Jacobson, 2005; Seinfeld and Pandis, 2006; Sportisse, 2009):

1. Eulerian models: These models can handle all emission sources over domains ranging from an urban area to the entire globe. Eulerian models consider a three-dimensional (3D) array of volume elements, each of homogeneous properties for the variables of interest (e.g., air pollutant concentrations, meteorological variables) and calculate the flow of those variables among those volume elements. All atmospheric processes can be modeled in Eulerian models (emissions, transport, transformations, and deposition). The outputs consist of the values of the variables (e.g., concentrations, deposition fluxes) as a function of time and location.

2. Lagrangian trajectory models: In a Lagrangian trajectory model, air masses are advected and pollutants are dispersed along mean wind trajectories. They are computationally less demanding than Eulerian models, but, unlike Eulerian models, Lagrangian models are typically limited to the simulation of a few sources.

3. Plume-in-grid models: A plume-in-grid model combines the advantages of Eulerian models (large domain and multiple sources) and Lagrangian models (fine resolution near the sources); these models simulate selected sources using a Lagrangian model imbedded within the 3D Eulerian model and all other sources are treated by the Eulerian model.

4. Gaussian dispersion models: Gaussian dispersion models are widely used to calculate pollutant concentrations downwind from a few selected sources in the absence of major obstacles to atmospheric transport. Stationary atmospheric conditions are assumed in the Gaussian plume dispersion formulation and, as a result, the impact of a source can be represented by a Gaussian plume model only over limited distances from the source (at most 50 km). When atmospheric conditions are variable, the atmospheric plume may be approximated by releasing distinct puffs from the source at successive intervals of time; the puffs may then follow different air mass trajectories.

5. Street-canyon models: For situations where the atmospheric dispersion of pollutants is constrained by obstacles such as buildings, street-canyon models must be used; they use parameterizations to approximate the effect of those obstacles on the atmospheric flow and pollutant dispersion.

6. CFD (Computational Fluid Dynamics) models: CFD models provide detailed representations of the atmospheric flow and some also treat the physics and chemistry of air pollutant transformations. However, they are limited to local applications such as the impact of a single pollution source in complex terrain or in a built environment, where the flow characteristics are complex. These models provide accurate representations of the atmospheric flow but are computationally demanding.

The selection of an air quality model for a specific application depends mostly on the spatial scale and source types being considered. The concentrations of air pollutants can be simulated over large domains, ranging from urban to global scales, by Eulerian and Lagrangian models. At local scales, Gaussian dispersion models, street-canyon models or CFD models are typically used. Plume-in-grid models can cover all spatial scales.

5.3.2.4 Stormwater models

The models of urban hydrology may be classified in terms of their functionality, accessibility, water quantity and quality components included in the model and their temporal and spatial scales (Zoppou, 2001; Elliot et al., 2007). In terms of spatial distribution, hydrological models may be categorized as follows:

1. Lumped: A lumped model is based on spatial averaging of the input parameters over the catchment. Therefore, these models provide only outputs at the outlet of the catchment without an explicit consideration of spatial variability.
2. Semi-distributed: Semi-distributed models take into account the variability of land use and inflow into the drainage network by dividing a catchment into several

subcatchments. Therefore, the spatial resolution is related to the size and number of subcatchments.

3. Fully-distributed: A fully-distributed model represents the surface water flow by using physical laws over a gridded domain. They include spatial and temporal variability (such as soil properties, land use, etc.) according to the grid size.

5.3.3 Model Integration

The main building blocks of a modelling chain of the environmental impacts of road traffic are (1) the overall structure of the modelling chain, which should include traffic, emissions, air pollution, and water quality models (Figure 5.5) and (2) the interfaces linking the output from a model to the input of the next model. The selection of the models and these integrations are briefly discussed below.

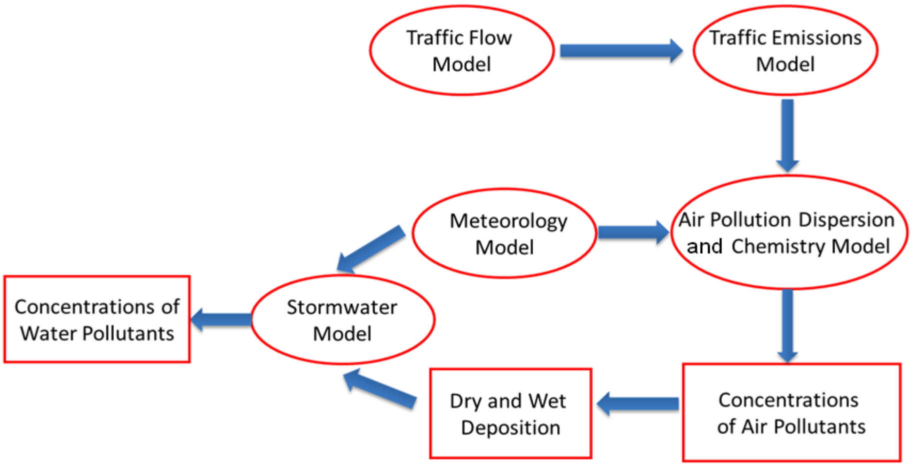

Figure 5.5. Schematic representation of the modelling chain (components are represented with ellipses) with expected input and output data (represented with boxes).

The selection of the models must be conducted in such a way that there is consistency among the level of detail, spatial and temporal resolutions, and input requirements of the various models of the modelling chain. The use of a dynamic traffic model will be of little use if the emission model is based on average vehicle speed. Similarly, an air quality model with a 50 km horizontal grid size will be inappropriate to investigate the impacts of local sources on an urban catchment. Once models that are internally consistent have been selected, the development of their interfaces will be mostly straightforward, because the inputs and outputs of the models should be compatible. There may, however, be some need for adapting the transfer of variables from one model to the next, because the original use intended for a model may differ from that considered in the modelling chain. For example, a traffic model designed to investigate congestion scenarios will differentiate among heavy-duty vehicles, light-duty vehicles, and passenger vehicles. Its use for an environmental impact study will require additional information on the types of vehicles present in each category, for example, diesel versus spark-ignition vehicles, as well as the vehicle original year (which defines the associated emission control characteristics and, therefore, pollutant emission rates).

In the case study presented below, we investigate the contribution of local road traffic to the contamination of stormwater runoff in an urban catchment. Since we are interested in the impact of traffic on major roads with relatively smooth traffic conditions, a static traffic model (category 1 in section 5.3.2.1) may be sufficient, because the impact of acceleration and deceleration on pollutant emissions may not be as critical as it would be, for example, at a major intersection. Then, an emission model based on vehicle speed would be appropriate (category 3 in section 5.3.2.2). Because we are focussing on the impacts of local traffic, it is important to resolve the atmospheric concentrations and deposition fluxes of pollutants at a fine spatial scale near the roadways. Gaussian models for line sources are appropriate for this type of air pollution impact study (category 4 in section 5.3.2.3). Finally, the water quantity and quality model must be consistent with the output of the atmospheric model and the objective of the study (contamination of stormwater at the catchment outlet). Thus, a semi-distributed model (category 2 in section 5.3.2.4) will provide some level of detail in terms of subcatchment variability, without imposing unneeded computational and input data burden on the study. Clearly, other choices could be made and the user's judgment is an important part of the construction of a modelling chain. We present in the following section the specific models selected for the study of the Grigny urban catchment and the results of the model simulations.

5.3.4 Case Study: Design, Methodology, and Results

The objective of this study is to simulate the impact of road traffic on stormwater contamination, to compare the modelling results to available measurements, and to investigate the sensitivity of those results to the level of detail of traffic air pollutant emissions. The present study focuses on three metals emitted by road traffic: cadmium (Cd), lead (Pb), and zinc (Zn). The Grigny catchment is located 20 km south of Paris in a suburban area. The catchment area is 451 ha, covered by several municipalities and divided into 32 subcatchments according to the topography and the structure of the drainage system. The geographical location and division of the catchment are presented in Figure 5.6. The models, available data and results are presented sequentially for each component (traffic, emissions, atmospheric pollution, stormwater contamination).

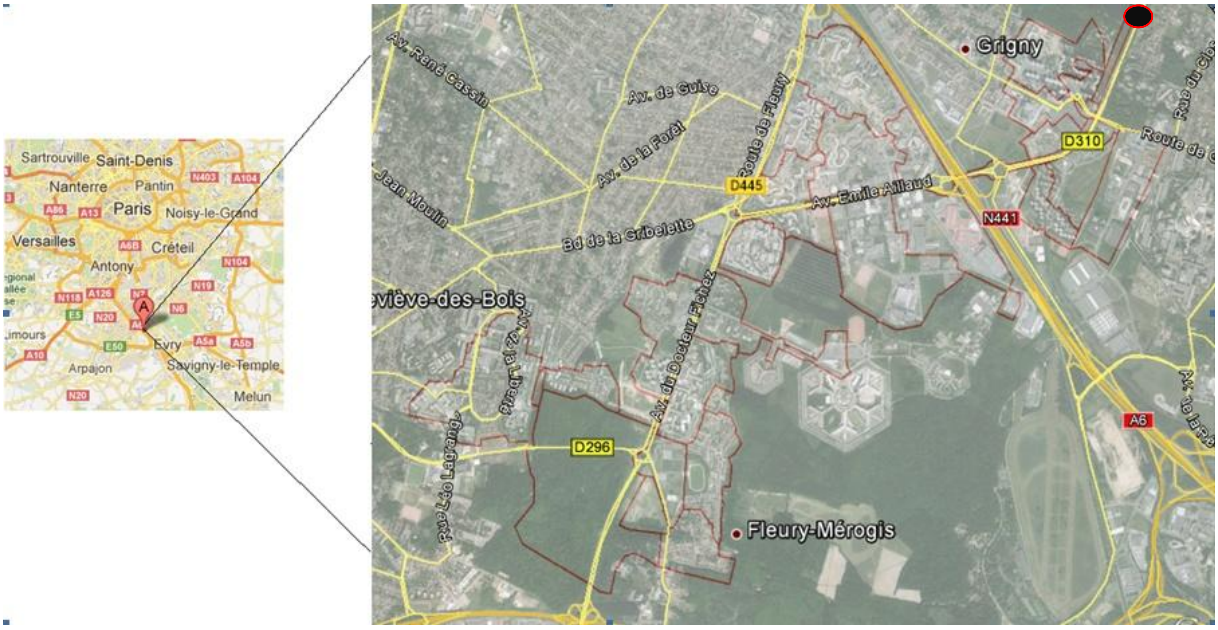

Figure 5.6. Geographical location and characteristics of the Grigny catchment including major roads (in yellow), outlet (black circle), and subcatchment boundaries (in red). (Source: Google Maps)

Traffic

The Grigny catchment is impacted by four main roads (D310, D296, N441, and D445) and the A6 freeway. Based on traffic variability and road direction, these roads were divided into eleven sections assumed to be of uniform traffic flow, speed, and fleet composition.

Annual average hourly traffic volumes on the A6 freeway were obtained from traffic count equipment operated by the French Ministry of Ecology, Sustainable Development, and Energy. Traffic volumes of the other main roads were simulated using the VISUM traffic model by the Regional Department of Transportation and Planning (DRIEA) (Figure 5.7). Their hourly traffic volumes were assumed to follow the hourly distribution of the A6 freeway traffic observations.

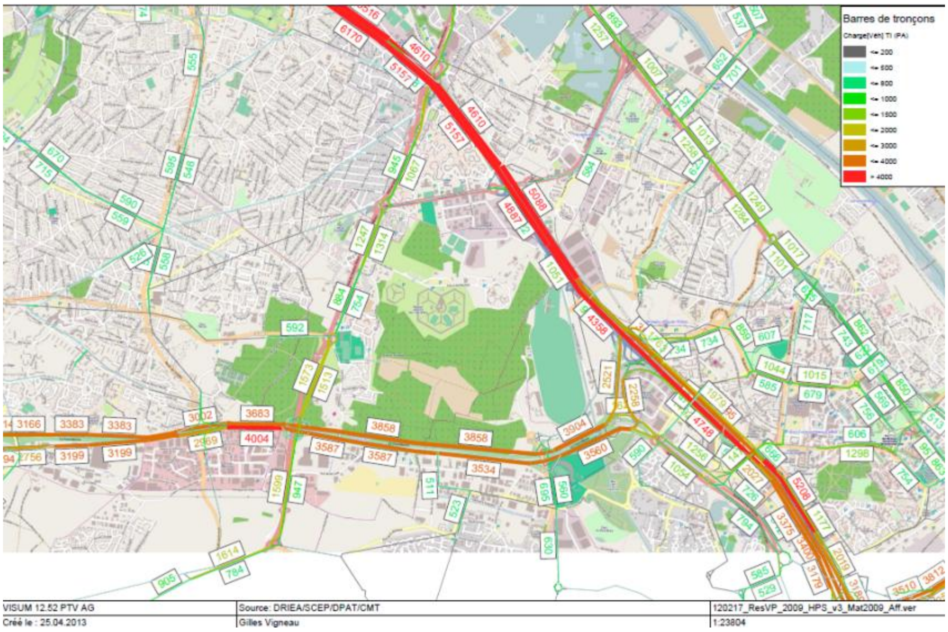

Figure 5.7. Peak hour traffic volumes in the Grigny catchment area (source: DRIEA).

The local traffic flow on residential streets was treated as a uniform area source based on the number of workers who live and commute out of the Grigny catchment area and the number of vehicles registered in the area. Such data were obtained from the French National Institute for Statistics and Economic Studies (INSEE). Traffic flow on residential streets accounts for 1% of total traffic in the study area.

Traffic emissions

Since the traffic output data are hourly, pollutant emissions could be estimated by an average-speed based model. For this purpose, the CopCETE model of the French Ministry of Ecology was used for exhaust emissions. It is based on the COPERT4 methodology (Ntziachristos et al. 2009). This model calculates vehicle emissions from most processes including vehicle exhaust, fuel evaporation, and equipment wear depending on road gradient, length of road, different types of traffic density (urban or rural road), fleet composition, number of vehicles per categories (passenger, light-duty, and heavy-duty vehicles, buses) and their average speeds. The fleet composition was assumed to be the same as the urban fleet composition for surface roads and main roads and a typical freeway fleet composition for the A6 freeway (Hugrel et al., 2004). Road dust contains particles derived from a wide range of sources. They can be classified into 4 categories: 1) vehicle sources (exhaust particles, tyre, break and clutch wear), 2) roadway sources (road surface wear, corrosion of crash barrier), 3) non-transport sources (e.g., industrial and commercial activities, vegetative detritus), 4) atmospheric deposition derived from the above sources in other areas. Legret and Pagotto (1999) found road dust to be heavily polluted by Pb, Cu, Cd, and Zn, originating from traffic. Currently, because of emission control regulations, exhaust emissions from road traffic have been reduced, but non-exhaust emissions from road vehicles are still largely unabated (Thorpe and Harrison 2008). For example, the ban of leaded gasoline significantly decreased Pb emissions from vehicles. In this study, non-exhaust emissions were calculated with the COPERT4 methodology for road vehicle tyre and brake wear, but road surface wear was not included due to a lack of data. A significant fraction of metals emitted by equipment wear may be present in coarse particles; therefore, emissions of particles greater than 10 µm in aerodynamic diameter are taken into account in the emission inventory used here. Another source of uncertainly is that re-emission of road dust by traffic was not taken into account; although this process may account for a significant fraction of atmospheric particulate matter (PM) in urban areas (e.g., Pay et al., 2011), it is still poorly characterized and, therefore, highly uncertain. An example of output from the emission model is provided in Figure 5.8.

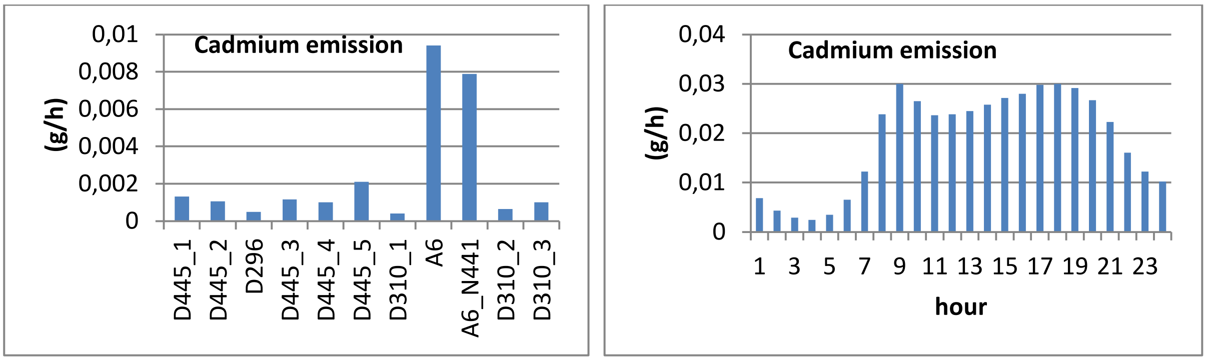

Figure 5.8.Cd emissions (g/h) for each road section at 10 am (left) and hourly profile of daily Cd emissions due to road traffic in the modelled area (right).

Atmospheric dispersion

The dispersion of atmospheric pollutants was calculated using the Polyphemus Gaussian dispersion model for a line source (Briant et al., 2011, 2013). The Gaussian dispersion formulation for a line source is exact to simulate emissions from road traffic when the wind is perpendicular to the line source and approximations are needed for other wind directions. The Polyphemus model uses a novel method to reduce the calculation error when the wind direction is not perpendicular to the road.

The model was used to simulate pollutant dispersion from major roadways in the catchment for the period from 2 April to 3 May 2012. The dataset used contains the following:

1. The coordinates of 4 roads and 1 freeway divided into 11 sections representing a total of 10 km of linear road length.
2. The Cd, Zn, and Pb hourly emission rates associated with traffic for each one of the 11 sections computed with the CopCETE emission model for exhaust emissions and the COPERT4 database for non-exhaust emissions.
3. The receptor locations; the overall receptor grid consists of several rectangular grids of 50 m spacing assigned to each one of the subcatchments.
4. Meteorological data required by the Polyphemus Gaussian model, which include wind speed and direction and cloud coverage. Meteorological variables are based on observations at the nearest measurement station (Orly airport, 8 km from the catchment). The use of the wind speed and direction from a close but different location is appropriate in this case because of the relatively flat terrain.

Figure 5.9 shows the modelled Zn concentrations (µg/m^3) at ground level on 5 April at 9 am. Only the concentrations located within the catchment are shown. These values correspond solely to traffic emissions, i.e., without background atmospheric concentrations. The concentrations decrease when the distance of the receptor from the road increases. The concentration at each point varies in time depending on traffic and meteorology.

In addition to the emissions associated with traffic from the major roadways, the contribution of emissions associated with local traffic on residential streets and that of background concentrations were also taken into account. The contribution of traffic flow on residential streets was simulated using a box model approach, which implies that those emissions are spatially uniform over the study area. We assumed that advection processes are more important than convection for air pollutant dispersion; therefore, the atmospheric concentrations associated with those emissions are inversely proportional to the wind speed (Benarie, 1967).

Background concentrations were obtained from measurements conducted by Ayrault et al. (2010) near Paris. The average urban background total suspended particulate (TSP) values are 0.393, 15.37 and 45.8 ng/m^3 for Cd, Pb, and Zn, respectively. These background concentrations were assumed to be spatially uniform over the study area and constant in time.

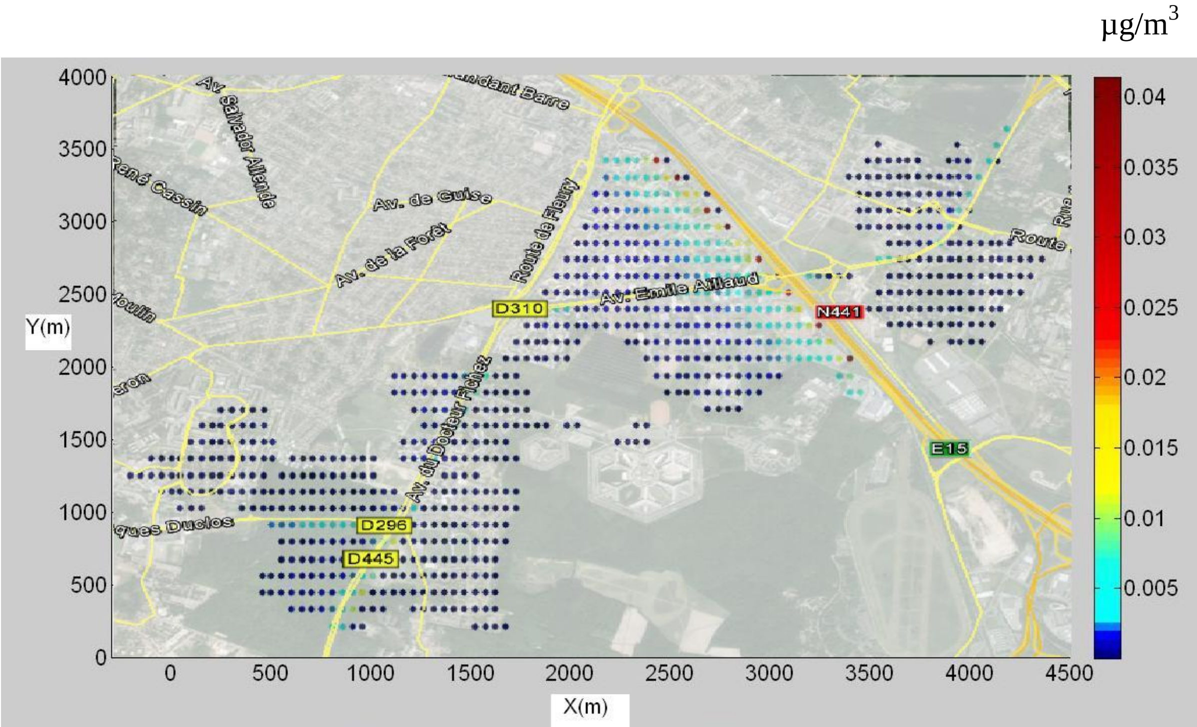

Figure 5.9. Zn concentrations (µg/m^3) due to traffic on major roads over the Grigny catchment on 5 April at 9 am on a receptor grid (m) corresponding to the urban catchment. The southwestern origin is at 48° 39’ 41’’ N, 2° 20’ 02’’ E.

Atmospheric deposition

Atmospheric deposition occurs via dry processes (i.e., when gas molecules and aerosol particles get into contact with surfaces) and wet processes (i.e., when gases and particles are scavenged by precipitation, mostly by rain). Several studies have experimentally quantified atmospheric deposition of metals near roadways (Viard et al., (2004); Azimi et al., (2005);

Sabin et al., (2006); Loubet et al., (2010)). These studies show that deposition fluxes decrease rapidly as the distance from the roadway increases, which is consistent with the spatial gradient observed for atmospheric concentrations. Theoretical models have been developed for dry deposition (e.g., Zhang et al. (2001), Wesely (2007), Sportisse (2007)) and wet deposition (e.g., Duhanyan and Roustan, 2011). The model formulations presented below follow those of standard theoretical models.

The dry deposition flux is typically computed using the following formulation:

$$F_{dry}(x,y) = V_d . C(x,y,z) \tag{5.3}$$

where V_d is the dry deposition velocity, which is a function of meteorology, land use, and particle size. The deposition velocities used here were obtained from Roustan (2005). The size distribution of PM_{10} mass in the atmospheric background was assumed to be 74% in particles with a diameter less than 1 µm, 10% in particles between 1 and 2.5 µm, and 16% in particles with a diameter between 2.5 and 10 µm. The size distribution of particles from traffic emissions consists of 58% of particles with a diameter less than 1 µm, 8% of particles between 1 and 2.5 µm, 13% of particles between 2.5 and 10 µm, and 21% of particles greater than 10 µm. The coarse particles (>10 µm) results mostly from non-exhaust processes. The selected values for the dry deposition velocities of particles in these four size ranges were 0.2, 0.4, 4 and 20 cm/s, respectively. The subcatchments were assumed to have approximately the same land use. The effect of meteorology on dry deposition velocities was not taken into account here and time-average values were used.

Pollutant buildup on the subcatchment surfaces is calculated according to the dry deposition flux of each pollutant source (i.e., local traffic, background concentration, traffic on the roads and the freeway), the associated level of traffic and the meteorological situation. 1784 virtual receptors separated by a distance of 50 m from each other and covering the whole catchment were selected (Figure 5.8). The average deposition over all receptors located in each subcatchment determines the buildup for each hour.

Wet deposition was assumed to be homogeneous over the catchment. The atmosphere over the catchment was assumed to be a well-mixed box with a height of 1000 m corresponding to a typical mixing height. The following formulation then applies.

$$F_{wet} = \frac{E_{total} . \Lambda}{\sqrt{S} . u} \tag{5.4}$$

where, E, S, u and Λ are the total emission of the pollutant within the domain, the surface of the catchment, the wind speed, and the wet deposition scavenging coefficient, respectively. Λ was calculated following the Andronache (2004) formulation for urban areas.

$$\Lambda = 6.67 \times 10^{-5} I^{0.7} \tag{5.5}$$

where I is the rain intensity (mm/h). The average pollutant concentration in rainfall can be calculated as the wet deposition flux divided by the precipitation amount over the rain event.

Water quality

The water quality simulation requires a satisfactory water quantity simulation because pollutant concentrations are highly dependent on the simulated flow rate. The model used to perform the stormwater runoff analysis is SWMM 5 (Rossman, 2010). This model is an open-source modelling software, well adapted to this investigation as it allows rainfall-runoff simulations (quantity and quality) over long periods with short time steps (2 min in this case). The water quantity modelling for the Grigny catchment is performed using land use and topography data, and is completed by a calibration with a genetic algorithm. The performance of the model is evaluated by the Nash-Sutcliffe criterion. Details on the model, its setup for the Grigny catchment and the calibration procedure followed are presented by Petrucci *et al.* (2013).

Water data availability and treatments

Continuous measurements of flow rate and turbidity were conducted by SIVOA (Syndicat mixte de la Vallée de l'Orge Aval) at the outlet of the Grigny catchment and are available for the whole year 2012. In this study, 14 rainfall events were selected from 2 April to 3 May 2012.

Continuous measurements were complemented by chemical analyses on several rainfall events. An automatic sampler measured the different metal concentrations (Cd, Zn, and Pb) of 6 distinct samples of a rainfall event occurring on 5 November 2012 (Tableau 5.1). For this rainfall event, the Cd concentration was below the quantification threshold for 4 measurements out of 6.

Tableau 5.1. Table 1. Metal concentrations on 5 November 2012 (SIVOA)

Time	Total Suspended Solid (TSS) (mg/l)	Cd* (µg/l)	Zn (mg/l)	Pb (µg/l)
17 h 44	210	<1	0,24	20,28
18 h 02	103	<1	0,21	14,57
19 h 02	223	1,08	0,26	24,83
20 h 02	191	1,04	0,18	18,58
21 h 02	126	<1	0,12	10,45
23 h 02	66	<1	0,1	9,61

(*). Half the quantification limit (i.e., 0.5 µg/l) was used when Cd concentrations were below the quantification limit.

The "observed" continuous metal concentrations were re-constructed over the period by calculating the total suspended solids (TSS) concentration using the TSS-turbidity relation and the ratio between TSS and metal concentrations obtained by the available samples. The averages of these ratios are 5.14×10^{-6}, 0.11×10^{-3} and 1.29×10^{-3} µg/l for Cd, Pb, and Zn, respectively. These ratios were used to calculate continuous metal concentrations at the outlet of the catchment.

Water quantity was simulated for 14 rainfall events. Half of the rainfall events (from 2 to 19 April) were used to calibrate the model and half (from 20 April to 3 May) were used for the model validation. Figure 5.10 compares the simulated flows with continuous measurements. The Nash values are 0.6772 and 0.6063 for calibration and validation, respectively.

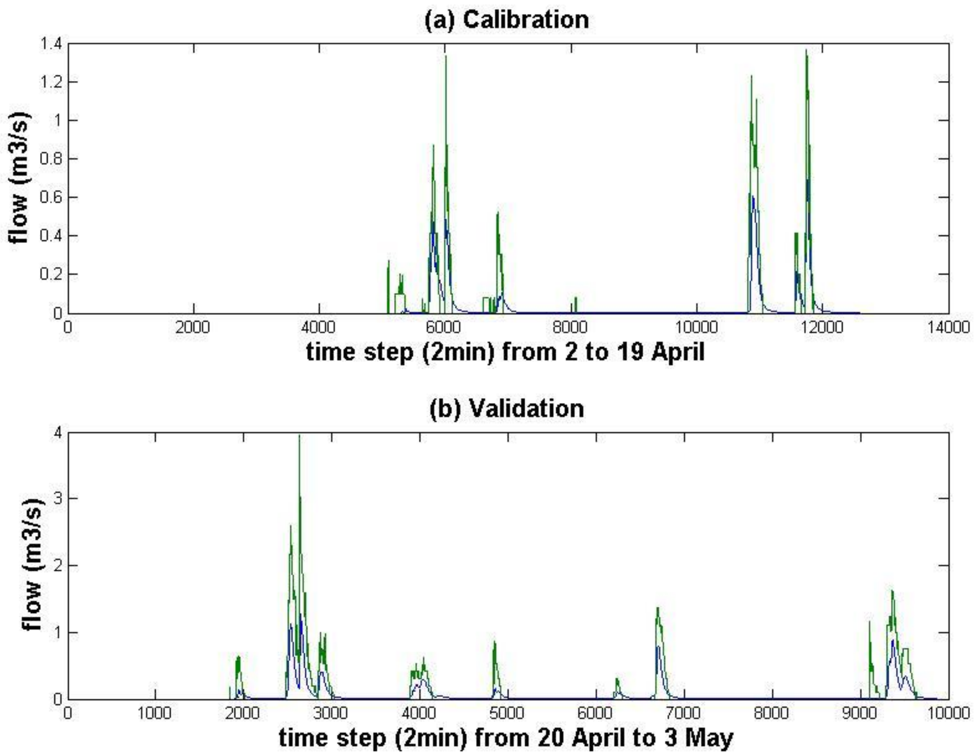

Figure 5.10. Comparison between simulation (blue line) and measurements (green line) results from 2 to 19 April (top, calibration) and from 20 April to 3 May (bottom, validation) at the outlet.

Water quality modelling

Usually water quality simulations include the pollutant buildup during dry periods and washoff during rainfall events. Different mathematical approaches are available to represent the processes governing pollutant accumulation and washoff. In this study, the buildup is obtained from the atmospheric deposition calculations and the power washoff equation can be written as follows:

$$W = E_1 q^{E_2} B \qquad 5.6$$

where, W is the washoff load (mass per hour), E_1 is the washoff coefficient, E_2 is the washoff exponent, q is the runoff rate per unit area (mm/hour) and B is the available mass of buildup. Thus, the pollutant washoff load is proportional to the product of runoff raised to some power and to the amount of available pollutants.

The washoff parameters, E_1 and E_2 depend on land use (Cheah 2009). On the basis of the available literature and without further information on the catchment, the chosen values for the parameters are constant average value: E_1=0.1, E_2=1.5.

Three model simulations were conducted:

(1) a reference model simulation including all emission sources and the background atmospheric concentrations, based on the model configuration described above

(2) a model simulation without any traffic emissions from the study area but including background atmospheric concentrations

(3) a model simulation with traffic emissions, but with emissions for traffic on major roadways treated as spatially-averaged over the study domain, which is defined as the smallest rectangular area covering the catchment and extending up to 1000 m from the surface (i.e., within the atmospheric mixing layer).

The results of the reference simulation can be compared to measured concentrations at the catchment outlet. A comparison between the first and second simulations allows us to investigate the contribution of local traffic within the catchment area to stormwater contamination. A comparison between the first and third simulations provides a quantitative assessment of the uncertainty associated with a spatially-averaged treatment of all emissions within the study area compared to the reference simulation where the emissions associated with major roadways are treated explicitly (i.e., spatially distributed).

Comparison of model simulation with measurements

The results presented in Figure 5.11 show the comparison of metal mass flow rates (µg/s) between the simulation and the measurements. The rainfall event of 24 April was excluded because the flow rate was not estimated correctly. The average ratios of modeled and measured mass flow rates are 77% for Zn, 100% for Cd, and 211% for Pb. These ratios vary among the fourteen rain events and range from 9 to 106% for Zn, from 11 to 150% for Cd, and from 24 to 296% for Pb. These results show that the impact of atmospheric deposition of Zn, Cd and Pb on water contamination is considerable. Zn originates from the corrosion of building materials, which can explain the fact that the simulation concentrations underestimate the measurements (Gromaire et al. 2011). Unleaded gasoline is now used in Europe, and Pb due non-exhaust emissions has also been reduced, but it still remains an important source of water contamination (Gromaire et al. 2011). In this study, the COPERT4 non-exhaust emission factor for Pb was used. Because this emission factor includes data between 1983 and 2007, it is likely to overestimate current Pb emissions because of reductions in Pb non-exhaust emissions in recent years. For example Hjortenkrans et al. (2007), showed that Pb emissions from brake linings in Stockholm (Sweden) were reduced from 560 kg/year in 1993 to 35 kg/year in 2005. The results for Cd should be taken with caution because two-thirds of the measurements were below the quantification limit and using half the quantification limit may over- or underestimate the actual contamination.

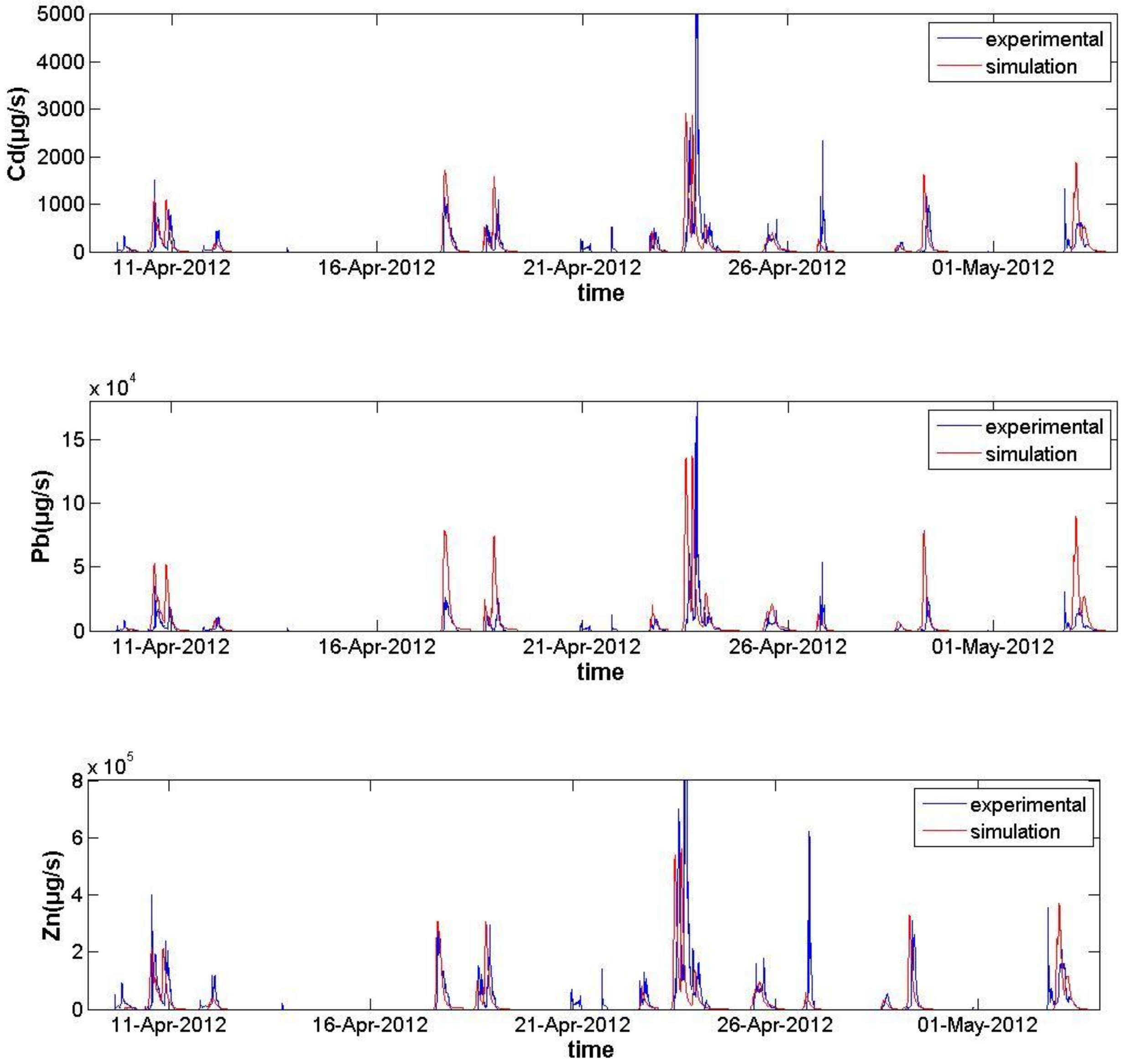

Figure 5.11.Metal mass flow rates (µg/s) at the outlet of the Grigny catchment: simulation results with an explicit treatment of traffic (in red) and experimental results (in blue). The rainfall event of 24 April was excluded (see text).

Impact of road traffic in the study area

The comparisons between the simulations with and without considering local traffic within the study area are presented in Figure 5.12. The comparison of these two cases (with and without traffic) demonstrates a significant effect of local traffic on Cd, Pb and Zn mass flow rates. The background contribution is 4% of the Zn concentrations on average (calculated as the average of the contributions for each rain event), with a range of 2 to 6% among the fourteen rain events. For Pb, the background contribution in stormwater is 6%, with a range of 3% to 8% among rain events. For Cd, the background contributes 7% in stormwater, with a range of 4% to 10% among rain events. The traffic contribution is dominant because (1) traffic emissions are close to the ground and, therefore, available for dry deposition and (2) the presence of particles with diameters greater than 10 µm from vehicle equipment wear (i.e., with large dry deposition velocities) leads to significant dry deposition near the roadways.

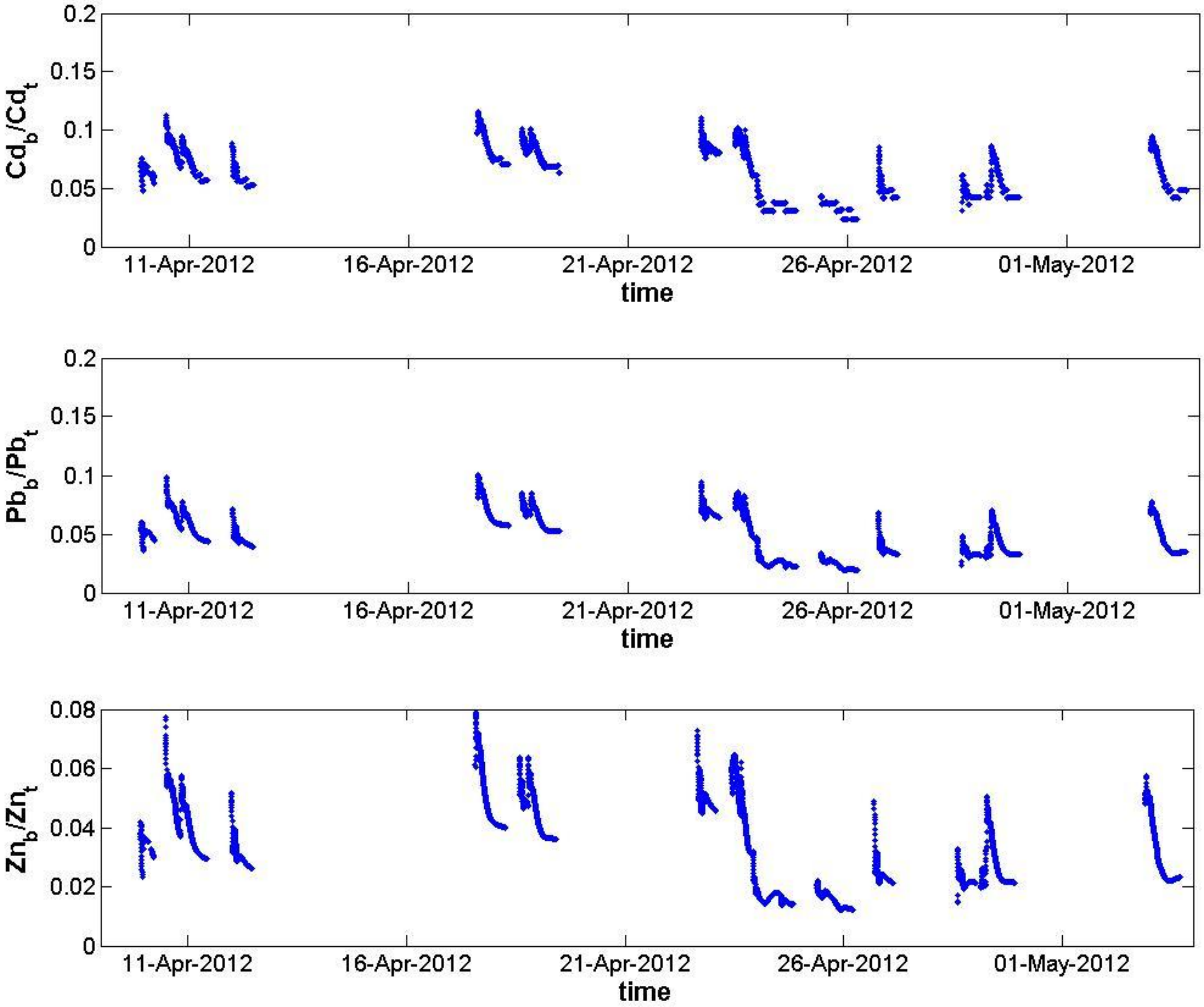

Figure 5.12. Cd, Pb, and Zn mass flow rate ratios between the case with an explicit treatment of local traffic ($metal_t$) and without local traffic ($metal_b$) in the study area at the outlet of the Grigny catchment. Only the average background atmospheric concentration was used in the second simulation.

Effect of the spatial representation of traffic on major roadways

The metal concentrations in runoff were simulated by using two different representations of the emissions associated with traffic on major roadways: in the reference simulation, four roadways and one freeway were represented explicitly, In the other simulation, the emissions from those major roadways were treated as those from residential streets, that is, they were spatially averaged and treated uniformly over the study area covering the catchment and their emissions were spatially distributed within a volume covering the study area with a height of 1000 m. This comparison provides, therefore, important information on the benefits of using an explicit representation of the emissions from major road traffic.

Figure 5.13 shows the temporal variation of the concentrations simulated with those two model configurations. The average ratios of the mass flow rates calculated by those two models (spatially-averaged / spatially-distributed) averaged over all rain events are 24% for Zn, 41% for Cd, and 34% for Pb. These ratios vary among the fourteen rain events and range from 11% to 38% for Zn, from 21 to 58% for Cd, and from 16 to 50% for Pb. The spatially-averaged results are low compared to those of the spatially-distributed model, because of the dilution of the traffic emissions within the atmospheric mixing layer over the study area. It means that averaging emissions with a box model leads to reduced impact of local traffic sources that have a strong impact on concentrations at the catchment outlet. Therefore, the differences in concentrations of metals emitted by traffic between the two simulations demonstrate that it is essential to have a correct spatial representation of those local emission sources.

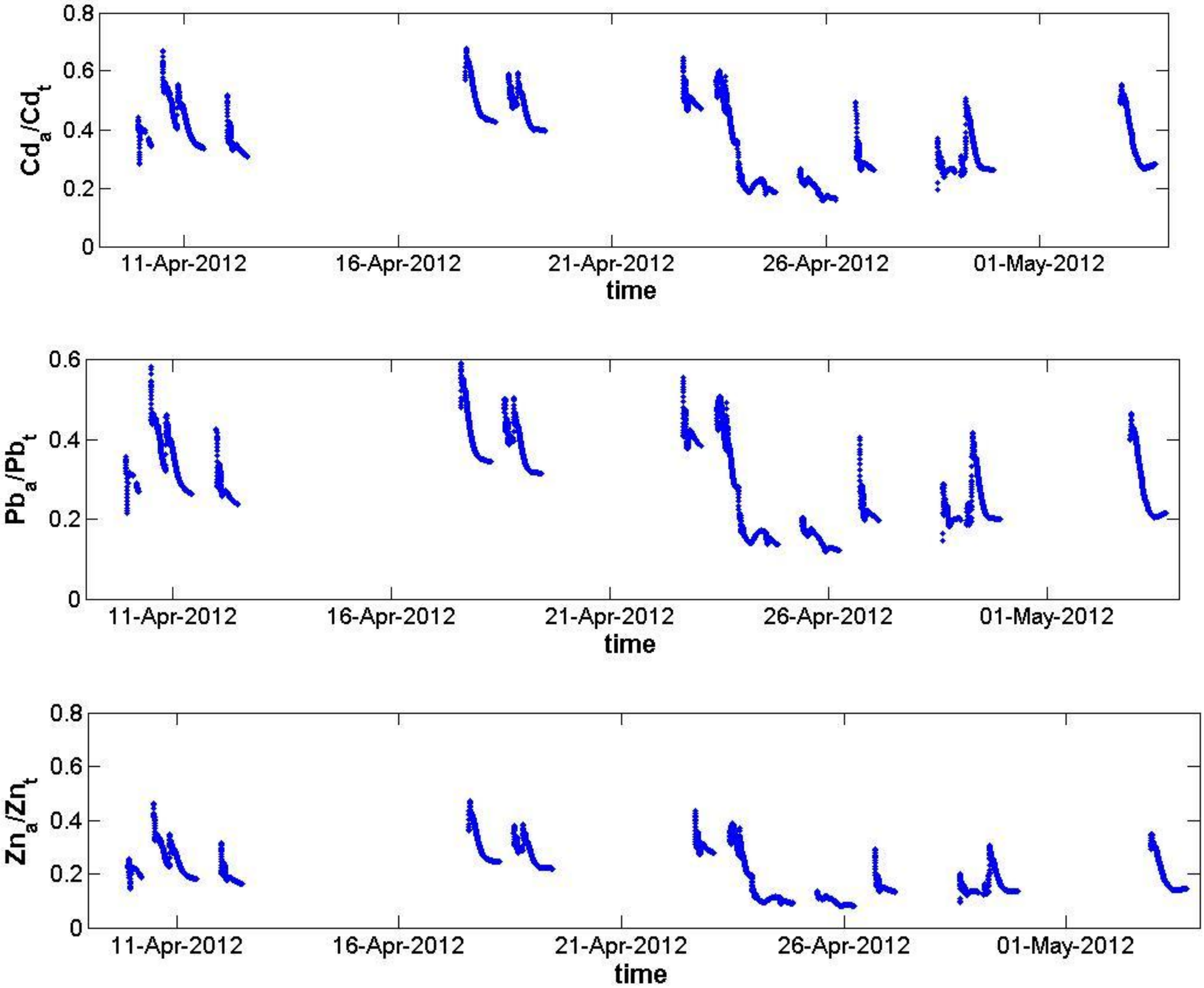

Figure 5.13. Ratios of Cd, Pb and Zn mass flow rates at the outlet of the Grigny catchment with spatially-distributed ($metal_t$) and spatially-averaged ($metal_a$) traffic emissions.

5.3.5 Discussion

This first step into air and water integrated quality modelling shows some limitations that should be overcome in the next few years to make these integrated models more reliable to predict contaminant concentrations in urban waters. First of all, the major drawback of this integrated model chain comes from the different modelling bricks themselves.

In order to simulate the environmental impacts of traffic at the local urban scale, it was necessary to choose the most appropriate modelling tools in terms of computational efficiency, which is a very important point when several models have to be coupled to constitute a modelling chain. Moreover, a major scientific stumbling block is that air quality models typically consider only those particles smaller than 10 µm in aerodynamic diameter (PM_{10}) since coarser particles are not regulated. This consideration is inherited from public health concerns and is related to the fact that only PM_{10} and fine particles affect human health. However, the mass of particles greater than 10 µm is significant and is relevant to water quality. Accordingly, it should be taken into account because particles of diameter above 10 µm are measured with the turbidimeter. Here, we took into account those coarse particles emitted by vehicle equipment wear; however, we did not account for road wear nor the re-suspension of road dust. Clearly, uncertainties are associated with every component of the modelling chain and some of those are discussed below.

For example, the most widely used air pollution models for calculating the dispersion of vehicle emissions at local scales are Gaussian dispersion models. However, in a densely built environment, the Gaussian dispersion assumption no longer applies and street-canyon models or CFD models may be more suitable. A comparison between Gaussian dispersion models and more detailed ones as part of a similar modelling chain is an interesting perspective. Furthermore, the background air pollution was treated here in a simple manner, using spatially uniform and temporally constant values in a box model. A plume-in-grid approach would provide a more comprehensive multi-scale modelling approach to better characterize this urban background in future studies.

Further, in an air-water modelling approach, the emission models must be able to estimate the main water pollutants such as total suspended solids (TSS), polycyclic aromatic hydrocarbons (PAH), and heavy metals (e.g., Pb, Zn, Cd, and Cu). It is a very restrictive need for emission models because most of them do not include these pollutants (in particular instantaneous models). Moreover, to consider a large number of pollutants and emission phenomena (e.g., including fuel evaporation and non-exhaust particle emissions), the microscopic emission models, which only treat a few pollutants emitted from the vehicle exhaust, have to be complemented with aggregated emission models such as COPERT4. However, those aggregated models pertain to large temporal scales, thereby leading to a discrepancy in the estimation of traffic emissions at very fine temporal scales.

At last, water quality can be well described by semi-distributed models such as SWMM, but the approach by subcatchments, adapted to hydrologic considerations, is not necessarily suited for coupling with the air component. The resolution used for the Grigny catchment (average subcatchment size) is about 200 m, coarser than that of the output from the deposition model. A fully-distributed hydrologic model would offer the advantage of explicitly taking into account distributed land uses and road surface characteristics at a higher resolution, in addition to rainfall characteristics at the scales of the resolution of the atmospheric receptors. Therefore, future work may use a fully distributed model for surface water flow and water quality modelling coupled to a hydraulic model for the urban drainage network.

The second major source of uncertainties in this integrated modelling approach is the lack of data covering the different components of this modelling chain.

For example, to assess the relationship between turbidity and metal concentrations, it would be necessary to consider more measurements of metal concentrations in water at the outlet of the catchment. In fact, the relations between turbidity, TSS and metal concentrations are not simple (Hannouche at al. 2011; Wust et al. 1994). These relations may depend, for instance, on pH, which was not measured for the samples available in this study. Therefore it would be necessary to obtain different samples in different seasons to confirm these results.

Another source of uncertainty in this modelling chain is the estimation of atmospheric deposition fluxes, both wet and dry, because those are strongly dependent on the particle size distributions. In particular, the dry deposition velocity still remains difficult to estimate as a function of particle size, atmospheric conditions and surface types (Roustan 2005). As this velocity may vary by an order of magnitude or more depending on the particle size, the influence on the simulated water concentrations is surely very high and further work is needed to determine more accurately this parameter accounting for particle size distributions and for the distribution of metals between the different sizes of particles.

5.3.6 Conclusion

In this study, we developed, applied, and discussed a modelling chain linking road traffic emissions to air and stormwater concentrations. Different models for traffic, emission, atmospheric dispersion and stormwater flow and contamination were presented. This brief review provided the context needed for the selection of the most appropriate models to be coupled in order to implement an integrated and efficient modelling system for a simulation of air and water quality in urban areas. This approach was tested on a urban catchment in the Paris region.

The effect of traffic impact in the Grigny catchment located in a southern suburb of Paris was studied from 2 April to 3 May 2012. The simulated Zn, Pb, and Cd concentrations in pollution peaks are commensurate with experimental results at the outlet, thereby suggesting the important contribution of atmospheric deposition to stormwater contamination in urban areas. Comparison of metal loading from atmospheric deposition under two hypotheses (with and without local traffic) indicates an important contribution of local traffic to Cd, Pb and Zn in urban catchments. Furthermore, an explicit representation of major roadways in the catchment area was shown to have a significant effect on simulated mass flow rates and concentrations.

Through this process, we observed several limitations in the simulation of all phenomena (traffic, emission, atmospheric dispersion and stormwater). The main challenges are associated with the consistency between inputs and outputs of the different models constituting the modelling chain, especially the available data, the scale of the problem and its complexity.

The method developed in this work appears very promising in the context of water quality modelling. Currently, pollutant loads are usually calculated from urban land use. Here, the atmospheric loads near heavy-traffic areas appear to be major contributions. Therefore, the traditional approach may need to be reconsidered depending on the configuration of major roadways within the catchment area. These encouraging results may be useful to test different urban scenarios at the time of the design and planning of new urban areas.

To that end, experimental data are needed to calibrate and /or evaluate the different components of the modelling chain. Although reasonable agreements were obtained here between model simulations results and measurements at the catchment outlet, more work is needed to refine this modelling approach and evaluate it over different settings. Furthermore, this work highlights the importance of developing interdisciplinary research programmes with strong interactions between modelling and experimental studies.