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USE OF SITE-SPECIFIC DATA TO ASSESS MONITORED NATURAL ATTENUATION AND REMEDIATION OF A CONTAMINATED SITE WITH LIFE CYCLE ASSESSMENT

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Ce mémoire intitulé:

USE OF SITE-SPECIFIC DATA TO ASSESS MONITORED
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CONTAMINATED SITE WITH LIFE CYCLE ASSESSMENT

présenté par: DITOR Michael
en vue de l'obtention du diplôme de: Maîtrise ès sciences appliquées
a été dûment accepté par le jury d'examen constitué de:

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Isaac, as far as you're concerned: I love you very much…but you're grounded.

RÉSUMÉ

Bien que l'Analyse du Cycle de Vie (ACV) soit utile pour évaluer la réhabilitation des sites contaminés en permettant d'évaluer les impacts secondaires des procédés de réhabilitation, l'évaluation des impacts primaires associés au site est problématique à cause de la nature générique des modèles d'évaluation des impacts du cycle de vie (ÉICV). L'intégration des données spécifiques au site étudié permet donc d'augmenter la pertinence de ce type d'ACV.

Une étude de cas ACV est présentée ici concernant un site contaminé industriel situé sur un bras de l'océan Pacifique appelé Puget Sound. Les options de réhabilitation suivantes sont comparées : l'atténuation naturelle et le dragage avec élimination dans une installation confinée (« les activités de réhabilitation »). IMPACT 2002+ est utilisée comme méthode ÉICV.

Afin de comparer et d'évaluer les différentes perspectives possibles pour l'évaluation des projets de réhabilitation des sites contaminés au moyen de l'ACV, la contamination sur le site est considérée successivement comme faisant partie intégrante de l'écosphère puis de la technosphère. Dans le cas où le site contaminé est considéré comme faisant partie de l'écosphère, la contamination sur le site est incluse dans l'inventaire du cycle de vie pour l'option d'atténuation naturelle. Pour la seconde perspective selon laquelle le site contaminé est intégré à la technosphère, les impacts causés par la contamination sur le site ne sont pas comptabilisés.

Pour l'option de l'atténuation naturelle, des données spécifique au site sont utilisées pour développer un modèle de bilan de masse et estimer l'évolution dans le temps des concentrations présentes dans la colonne d'eau et les couches sédimentaires, en tenant compte de processus tels que le débit, les marées, la dégradation, la diffusion et la

sédimentation. Les profils de concentration obtenus grâce à ce modèle sont utilisés pour estimer l'impact écotoxicologique local ainsi que la masse des émissions dans l'eau et dans l'air provenant du site contaminé. Pour ce projet, ce modèle est désigné comme « le modèle d'Atténuation Naturelle » (*Natural Attenuation Model*).

Les impacts dus aux émissions dans l'eau provenant du site contaminés sont évalués à l'aide de trois versions différentes du modèle multimédia de devenir et d'exposition IMPACT : Une version non-spatiale, une version spatialisée avec comme échelle de résolution géographique les bassins versants et une autre version de ce dernier modèle spatialisé comprenant l'ajout d'une zone locale paramétrée pour représenter le bras d'Océan Puget Sound. Pour ce projet, ces trois versions du modèle sont désignés comme : la version *Non-spatiale*, la version *Bassin Versant* et la version *Bassin Versant Modifié* (*Non-spatial, Watershed* and *Modified Watershed*, respectivement).

Les impacts dus aux émissions dans l'air provenant du site contaminé sont évalués à l'aide de la version non-spatiale du modèle et d'une version spatialisée qui considère un environnement urbain compris dans le plan spatial du bassin versant. Ces modèles sont désignés comme: la version *Non-spatiale* et la version *Boîte Urbaine* (*Non-spatial* and *Urban box*).

Les impacts dus à la contamination dans les sédiments sur le site sont évalués au moyen des facteurs d'effet écotoxicologiques des sédiments, calculés à partir des facteurs d'écotoxicité aquatique et des coefficients de partition sédiment-eau, selon la méthode de partition d'équilibre (*Equilibrium Partitioning Method*).

Le modèle d'Atténuation Naturelle prédit que le site contaminé va émettre approximativement 290,000 kg de métaux et 3,500 kg de contaminants organiques dans l'eau et 8.7 kg de mercure et 480 kg de contaminants organiques dans l'air.
L'advection est le processus le plus important pour la disparition des contaminants de

la colonne d'eau. Afin de savoir quel est le principal processus de disparition des différents contaminants présents, les cinétiques de ces processus sont prises en compte. Dans un premier temps la contribution de la dégradation à la disparition totale est estimée, par la suite on regarde quelle est la contribution relative de la re-suspension et l'enfouissement pour expliquer la disparition de ce qui n'a pas été dégradé. La diffusion n'est pas ici un processus important pour le transfert de masse. Après 100 ans, le modèle prévoit l'absence de contaminants dans les sédiments mixtes. La disparition totale des contaminants organiques dans les sédiments profonds au bout de 100 ans est également prédite par le modèle, à l'exception des BPC.

L'utilisation de la méthode *Equilibrium Partitioning* pour calculer les facteurs d'écotoxicité dans les sédiments aboutit à quelques inversions notables par rapport à l'écotoxicité aquatique dans le cas des substances hydrophobiques avec une toxicité aquatique élevée. La résultante faible obtenue en terme d'écotoxicité dans les sédiments pour ce type de contaminants est contradictoire avec les objectifs de qualité des sédiments fixés pour le site contaminé. Bien que les facteurs d'effet écotoxicologique pour les sédiments calculés selon la méthode *Equilibrium Partitioning* n'aient pas pu être validés dans la présente étude, il est recommandée ici d'user de précaution lors de leur application et de leur interprétation.

Peu de différences sont observées entre les facteurs de caractérisation pour l'écotoxicité aquatique obtenus à l'aide des versions *Non-spatiale* et *Bassin Versant* du modèle *IMPACT*. Par contre, des différences de un à deux ordres de grandeur existent entre les facteurs de caractérisation produits par les versions *Bassin Versant* et *Bassin Versant Modifié*. Ces différences sont principalement expliquées par la différence de temps de résidence de l'eau entre la zone d'émission et l'océan pour les différentes versions, notamment pour les métaux, qui présentent des diminutions significatives d'impact écotoxicologique lorsqu'ils sont émis dans des eaux qui atteignent plus rapidement l'océan, pour lequel aucun impact écotoxique n'est évalué selon le modèle IMPACT.

Peu de différences sont observées entre les facteurs de caractérisation pour la toxicité humaine obtenus à l'aide des modèles *Non-spatial* et *Bassin Versant*. Par contre, des différences de un à trois ordres de grandeur sont observées entre les facteurs de caractérisation pour la toxicité humaine produits par les versions *Bassin Versant* et *Bassin Versant Modifié*. Ces différences importantes sont attribuable d'une part à une différence de temps de résidence de l'eau entre les différentes versions du modèle et d'autre part au fait que l'eau soit ou non considérée dans le modèle en tant que source d'eau potable (notamment, l'eau de Puget Sound n'est pas utilisée comme eau potable).

Les facteurs de caractérisation écotoxicologiques pour les émissions dans l'air calculés à l'aide de la version du modèle *Boîte Urbaine* sont 1.92 à 61.7 fois plus élevés que ceux provenant de la version du modèle *Non Spatial*. Les différences de taux de précipitation et de taux de ruissellement sont identifiées comme les principales causes de ces différences.

Les facteurs de caractérisation pour la toxicité humaine pour les émissions dans l'air calculés à l'aide de la version du modèle *Boîte Urbaine* étaient 1.81 à 702 fois plus élevés que ceux provenant de la version *Non-spatiale*. Les différences de taux de production agricoles, particulièrement pour les produits exposés à l'air, sont identifiées comme les principales causes des différences dans les résultats des différentes versions du modèle.

Les impacts sur le site lui-même, associés à la contamination dans les sédiments sur une période de 100 ans, sont estimés à 8.86E+08 PAF-m^3-an, dont 96% causés par le zinc et le cuivre seuls. Les impacts écotoxicologiques hors site pour toutes les émissions à l'eau sur une période de 100 ans totalisent 1.17E+09, 1.55E+09 et 3.66E+06 PAF-m^3-an respectivement pour les modèles *Non-spatial*, *Bassin Versant* et *Bassin Versant Modifié*, dont 96% sont attribués au zinc et au cuivre. Les impacts écotoxicologiques hors site pour l'ensemble des émissions à l'air totalisent 1.91E+05 PAF-m^3-an avec le modèle *Boîte Urbaine* et 1.85E+04 PAF-m^3-an pour le modèle *Non-spatial*, le mercure étant responsable de 99% des impacts dans les deux cas.

Les impacts toxicologiques hors site pour l'ensemble des émissions à l'eau totalisent 6.76E+01, 1.02E+02, et 2.29E-01 DALY pour les modèles *Non-spatial*, *Bassin Versant* et *Bassin Versant Modifié*, respectivement. Les métaux sont responsable de la majorité de l'impact, dont le zinc est responsable de 77 à 82% selon le modèle utilisé. Les impacts toxicologiques hors site pour toutes les émissions à l'air totalisent 1.39E-02 DALY pour le modèle *Boîte Urbaine* et 7.09E-03 DALY pour le modèle *Non-spatial*, le mercure étant responsable de 89% et 97% de l'impact pour les modèles *Boîte Urbain* et *Non-spatial*, respectivement.

L'ACV des activités de réhabilitation indiquent les résultats suivants pour les différentes catégories de dommages : 8.59E+02 DALY pour la santé humaine, 1.64E+07 PDF-m^2-an pour la qualité des écosystèmes, 1.07E+09 kg CO_2 eq pour les changements climatiques et 3.51E+10 MJ pour l'utilisation des ressources. La grande majorité de ces impacts est causée par l´utilisation d'un géotextile de polypropylène dans le site d´élimination.

Comme mentionné plus haut, la spatialisation et l'introduction de données spécifiques au site a abouti à l'obtention de résultats significativement différents en termes de facteurs de caractérisation. Cependant, les conclusions qui peuvent ressortir de l´application de ces facteurs de caractérisation sont moins claires et la précaution est recommandée pour l´interprétation des résultats.

Le modèle d'atténuation naturelle et le modèle IMPACT sont sensibles à certains paramètres qui ont une variation naturelle de plusieurs ordres de grandeur, ce qui implique une incertitude significative sur les résultats de ces modèles. La quantification de cette incertitude ne fait pas l'objet de cette étude, cependant comme règle générale, des différences de moins de deux ordres de grandeurs dans les résultats ACV ne peuvent pas être considérées comme significatives.

Quelles que soient la perspective adoptée et la version du modèle IMPACT utilisées, les activités de réhabilitation causent des impacts plus importants que l'atténuation naturelle pour les catégories de dommages « Changements Climatiques » et « Utilisation des Ressources ». Selon la perspective considérant que le site fait partie de l´écosphère, on ne constate pas de différence significative entre les deux options de gestion pour les catégories de dommages « Santé Humaine » et « Qualité des Écosystèmes », sauf lorsque les facteurs de caractérisation sont calculés avec la version *Bassin Versant Modifié* du modèle IMPACT. Dans ce dernier cas, les impacts de l'atténuation naturelle pour la catégorie « Santé Humaine » deviennent significativement moins importants que ceux des activités de réhabilitation.

Selon la perspective considérant que le site fait partie de la technosphère, le résultat est similaire, sauf que les impacts sur la Santé Humaine et sur la Qualité des Écosystèmes de l'atténuation naturelle sont significativement moindre que ceux des activités de réhabilitation lorsque l'on utilise la version *Bassin Versant Modifié* de IMPACT pour calculer les facteurs de caractérisation.

Les résultats indiquent que l´intégration de données spécifiques au site en ÉICV via la modélisation spatialisée de devenir et d'exposition peut modifier les conclusions d'une ACV comparant les différentes options de réhabilitation d'un site contaminé. Par ailleurs, une nécessité importante d'améliorer la modélisation de devenir des métaux a été constatée.

ABSTRACT

Contaminated site management requires a holistic approach that ensures that a net positive environmental benefit will be gained from remedial action. Such an approach evaluates both the local primary impacts caused by the site-related contamination and the secondary impacts associated with the remediation activities themselves. Life cycle assessment (LCA) has been used successfully to evaluate secondary impacts, however site-related impacts are poorly assessed due to the generic nature of life cycle impact assessment (LCIA) models. The integration of site-specific environmental data within LCIA has the potential for improving LCA's ability to assess contaminated site remediation options.

An industrial port in Puget Sound with contaminated sediments was taken as a LCA case study, comparing the following remediation options: monitored natural attenuation (MNA) and dredging with confined disposal ("the Remediation Activities"). IMPACT 2002+ was used as the LCIA method.

In order to demonstrate possible perspectives that can be taken when evaluating contaminated site remediation projects with LCA, the on-site contamination was evaluated alternately as part of the ecosphere and part of the technosphere. Using the "ecosphere perspective", the on-site contamination is included as part of the MNA life cycle inventory. Using the "technosphere perspective", the impact caused by the on-site contamination is not evaluated.

For the MNA option, site-specific data were used to develop a mass balance model and estimate concentrations over time for the water column and sediment layers, considering processes such as tidal flow, degradation, diffusion and settling. These

concentration profiles were used to assess the local ecotoxicity impact as well as the mass of water and air emissions from the contaminated site.

Impacts due to water emissions from the contaminated site were evaluated using three versions of the multimedia fate and exposure model IMPACT: a non-spatial version, a version spatially differentiated at the watershed level and a second version of the latter model with an added zone taking site-specific parameter values to represent Puget Sound. These models are referred to as the Non-spatial, Watershed and Modified Watershed model versions.

Impacts due to air emissions from the contaminated site were evaluated using the non-spatially differentiated model and a spatially differentiated model that considers an urban environment nested within the watershed spatial plan. These models are referred to as the Non-spatial and Urban box models.

Impacts due to on-site sediment contamination were evaluated using sediment ecotoxicty effect factors, calculated using aquatic ecotoxicity effect factors and sediment-water partitioning coefficients according to the Equilibrium Partitioning method.

The MNA mass balance model predicted that the contaminated site would emit approximately 290,000 kg of metals and 3,500 kg of organic contaminants to water and 8.7 kg of mercury and 480 kg of organic contaminants to air. Advection was found to be a dominant process in removing contaminants from the water column. Removal from the mixed sediments was best described by decay, with the fate of the remaining contaminants being split between resuspension and burial according to the velocities of these processes. Diffusion was not an important mass transfer process. After 100 years, no contaminants were left in the mixed sediments. No organic contaminants were left in the deep sediments after 100 years, with the exception of PCB.

The use of the Equilibrium Partitioning method to calculate sediment ecotoxicity factors resulted in some noteworthy inversions with respect to aquatic ecotoxicity in the case of several hydrophobic compounds with high aquatic toxicity. The corresponding low sediment ecotoxicity did not correlate with the sediment quality objectives set for the contaminated site. While the sediment ecotoxicity effect factors calculated using the Equilibrium Partitioning method were not validated in the present study, it is nonetheless concluded that caution should be used in their application and interpretation.

Little difference was observed between the aquatic ecotoxicity characterization factors produced by the Non-spatial and Watershed models, but a difference of one or two orders of magnitude was observed between ecotoxicity characterization factors produced by the Watershed and Modified Watershed models. Differences in water residence time between the emission zone and the ocean were responsible for characterization factor differences between the models, with metals showing significant decreases in ecotoxic impact when emitted into water that takes less time to reach the ocean where no ecotoxic impact is evaluated.

Little difference was observed between the human toxicity characterization factors produced by the Non-Spatial model and the Watershed model, however a difference of one to three orders of magnitude difference was observed between human toxicity characterization factors produced by the Watershed version and the Modified Watershed version. Residence time in water and whether the body of water was modeled as a drinking water source were responsible for the differences between models.

Ecotoxicity characterization factors for air emissions within the Urban box model were calculated to be 1.92 to 61.7 times higher than those for an emission to air within the

Non-Spatial model. Differences in rainfall rates and runoff rates were identified as the main drivers for differences between the model results.

Human toxicity characterization factors due to an emission to air within the Urban box model were calculated to be 1.81 to 702 times higher than those for an emission to air within the Non-Spatial model. Differences in agricultural production rates between the models, most notably for above-ground produce, were identified as causing the differences in model results.

The on-site impacts associated with contamination in the sediments over a 100-year time period was estimated at 8.86E+08 PAF-m^3-year, with 96% of the on-site impact being caused by zinc and copper alone. Off-site ecotoxicological impacts for all water emissions over the 100-year time period totalled 1.17E+09, 1.55E+09 and 3.66E+06 PAF-m^3-year for the Non-spatial, Watershed and Modified Watershed models, respectively, again with 96% being attributed to zinc and copper. Off-site ecotoxicological impacts for air emissions totalled 1.91E+05 PAF-m^3-year for the Urban box model and 1.85E+04 PAF-m^3-year for the Non-spatial model, with mercury responsible for 99% of the impact in both cases.

Off-site toxicity impacts for water emissions totalled 6.76E+01, 1.02E+02, and 2.29E-01 DALY for the Non-spatial, Watershed and Modified Watershed models, respectively. Metals were responsible for the bulk of the off-site ecotoxicological impact, with zinc alone being responable for 77 to 82%, depending on the model. Off-site toxicity impacts for air emissions totalled 1.39E-02 DALY for the Urban box model and 7.09E-03 DALY for the Non-spatial model, with mercury responsible for 89% and 97% of these impacts for the Urban and Non-spatial models, respectively.

The LCA of the Remediation Activities showed the following damage impact category results: 8.59E+02 DALY for Human Health, 1.64E+07 PDF-m^2-year for Ecosystem

Quality, 1.07E+09 kg CO_2 eq for Climate Change and 3.51E+10 MJ for Resources. The majority of this impact was caused by the use of a polypropylene geotextile liner at the disposal site.

As noted above, the IMPACT model results showed consistent differences between the characterization factors produced by the various model versions (with the exception of aquatic ecotoxicity characterization factors between the Non-spatial and Watershed model), due to the effect of spatial differentiation and the introduction of site-specific data. However, differences in the conclusions drawn from the application of these characterization factors are less clear and caution is required in interpreting the results.

Both the MNA and IMPACT models are sensitive to parameters that show natural variation across several orders of magnitude, indicating model uncertainty that spans a similar range. It is beyond the scope of this study to quantify this uncertainty, however as a rule-of-thumb, differences of less than two orders of magnitude in LCA results were not considered to be significant.

In all cases, the Remediation Activities cause greater impacts than MNA for the damage indicators Climate Change and Resource Use. Taking the ecosphere perspective, no significant difference in Human Health or Ecosystem Quality impacts can be noted between the management options, except where MNA is evaluated using the Modified Watershed model, in which case the MNA Human Health impacts are less than those of the Remediation Activities. Taking the technosphere perspective, the result is the same, except both the Human Health and Ecosystem Quality impacts of the Modified Watershed MNA are less than those of the Remediation Activities.

The results indicated that the integration of site-specific data into LCIA via spatially differentiated fate modeling can change the conclusions of a LCA comparing site remediation options. A particular need for improvement in the fate modeling of metals was observed.

CONDENSÉ EN FRANÇAIS

Introduction et revue de la littérature

L'Analyse du Cycle de Vie (ACV) est une analyse holistique qui évalue les impacts environnementaux associés à un produit, à un service ou à un procédé tout au long de son cycle de vie, de l'extraction des matières premières jusqu'à son élimination finale. Bien que l'ACV soit utile pour évaluer la réhabilitation des sites contaminés en permettant d'évaluer les *impacts secondaires* des procédés de réhabilitation, l'évaluation des *impacts primaires* associés au site est problématique à cause de la nature générique des modèles d'évaluation des impacts du cycle de vie (ÉICV). L'intégration de données spécifiques au site étudié permet donc d'augmenter la pertinence de ce type d'ACV. L'ÉICV évalue les impacts par le biais des facteurs de caractérisation, qui sont le produit des facteurs de devenir et des facteurs d'effet. Ce projet vise à intégrer les données spécifiques au site pour l'évaluation du devenir des substances.

Hypothèse

Les conclusions d'une analyse du cycle de vie comparant les options de réhabilitation d'un site contaminé peuvent être modifiées selon le niveau de régionalisation utilisé dans la méthode d'évaluation des impacts du cycle de vie.

Objectifs

1. Développer un moyen d'évaluer l'inventaire du cycle de vie de l'utilisation de l'atténuation naturelle comme option de gestion d'un bassin de marée contenant

des sédiments contaminés, en tenant compte de la contamination trouvée sur le site ainsi que des émissions hors site.

2. Intégrer des données spécifiques au site dans une méthode d'évaluation des impacts du cycle de vie afin de mieux représenter l'environnement local autour d'un site contaminé et d'évaluer les émissions hors site.

3. Comparer deux options de réhabilitation pour un site contaminé au moyen de l'analyse du cycle de vie, la première option présentant des impacts primaires significatifs et peu d'impacts secondaires, la deuxième peu d'impacts primaires et des impacts secondaires importants.

Méthodologie

Une étude de cas ACV est réalisée ici pour évaluer les différentes options de gestion d'une voie d'eau contaminée pendant 50 ans. Le site, situé sur un bras de l'océan Pacifique appelé Puget Sound, a des sédiments contaminés aux hydrocarbures aromatiques polycycliques (HAP), aux biphényles polychlorés (BPC), et aux métaux. Les options de réhabilitation suivantes sont comparées : le dragage avec élimination dans une installation confinée (ce qui est intitulé « activités de réhabilitation » dans la suite du document) et l'atténuation naturelle.

L'évaluation des activités de réhabilitation prend en compte le dragage de la voie d'eau, le transport des sédiments contaminés vers une installation d'élimination confinée, ainsi que la construction d'une digue pour cette installation.

Les activités évaluées pour l'atténuation naturelle sont la collecte des échantillons pour le suivi et le transport des échantillons jusqu'à un laboratoire.

Pour l'option de l'atténuation naturelle, des données spécifique au site sont utilisées pour développer un modèle de bilan de masse et estimer l'évolution dans le temps des concentrations présentes dans la colonne d'eau et les couches sédimentaires, en tenant compte de processus tels que le débit, les marées, la dégradation, la diffusion et la sédimentation. Les profils de concentration obtenus grâce à ce modèle sont utilisés pour estimer l'impact écotoxicologique local ainsi que la masse des émissions dans l'eau et dans l'air provenant du site contaminé. Pour ce projet, ce modèle est désigné comme « le modèle d'Atténuation Naturelle » (*Natural Attenuation Model*).

La sensibilité du modèle d'atténuation naturelle à une réduction de 10% de différents paramètres est testée. La variation des variables de sortie du modèle résultante est évaluée en faisant varier les paramètres d'entrée suivants: la vitesse du vent, le contenu en carbone organique des sédiments et des solides en suspension, la quantité de solides en suspension, la porosité, le coefficient de partition carbone organique-eau, la constante de Henry, le taux de dégradation dans l'eau, le taux de dégradation dans les sédiments et le facteur de débit de retour.

Afin de comparer et d'évaluer les différentes perspectives possibles pour l'évaluation des projets de réhabilitation des sites contaminés au moyen de l'ACV, la contamination sur le site est considérée successivement comme faisant partie intégrante de l'écosphère puis de la technosphère. Dans le cas où le site contaminé est considéré comme faisant partie de l'écosphère, la contamination sur le site est incluse dans l'inventaire du cycle de vie pour l'option d'atténuation naturelle. Pour la seconde perspective selon laquelle le site contaminé est intégré à la technosphère, les impacts causés par la contamination sur le site ne sont pas comptabilisés.

IMPACT 2002+ a été utilisée comme méthode d'évaluation des impacts du cycle de vie. Les impacts dus à la contamination dans les sédiments sur le site sont évalués au moyen des facteurs d'effet écotoxicologiques dans les sédiments, calculés à partir des facteurs d'écotoxicité aquatique et des coefficients de partition sédiment-eau, selon la méthode de partition d'équilibre (*Equilibrium Partitioning Method*). Cette méthode prend pour hypothèse que l'eau interstitielle est la voie d'exposition dominante et que les organismes benthiques sont aussi sensibles aux contaminants que les organismes aquatiques. Habituellement, en évaluation de risque écotoxicologique, les impacts dans les sédiments ne peuvent pas être déduit uniquement à partir de données chimiques. Des bioessais et des données sur les communautés benthiques sont également utilisés pour déterminer si l'écosystème en contact avec les sédiments a été affecté (c'est ce que l'on appelle l'analyse en triade de la qualité des sédiments). Étant donné qu'une évaluation de risque écotoxicologique a déjà démontré qu'un impact significatif existait au niveau des sédiments du site étudié dans la présente étude, il semble important de comptabiliser cet impact dans la présente étude : ces facteurs d'effet sont appliqués, mais uniquement pour les contaminants présents au niveau du site. En l'absence d'évaluation de risque écotoxicologique disponible ailleurs que sur le site lui-même, il a été jugé inopportun d'appliquer ces facteurs pour les substances qui se retrouvent dans les sédiments en dehors du site étudié. Il est supposé que cette incohérence dans l'application des facteurs d'effet écotoxicologique des sédiments ne sous-estime pas de façon significative l'impact causé par l'option d'atténuation naturelle.

Les impacts dus aux émissions dans l'eau provenant du site contaminé sont évalués à l'aide de trois versions différentes du modèle multimédia de devenir et d'exposition IMPACT : Une version non-spatiale, une version spatialisée avec comme échelle de résolution géographique les bassins versants et une autre version de ce dernier modèle spatialisé comprenant l'ajout d'une zone locale paramétrée pour représenter le bras d'Océan Puget Sound. Pour ce projet, ces trois versions du modèle sont désignées comme : la version *Non-spatiale*, la version *Bassin Versant* et la version *Bassin Versant Modifié* (*Non-spatial*, *Watershed* and *Modified Watershed*, respectivement).

Les impacts dus aux émissions dans l'air provenant du site contaminé sont évalués à l'aide de la version non-spatiale du modèle et d'une version spatialisée qui considère un environnement urbain compris dans le plan spatial du bassin versant. Ces modèles sont désignés comme: la version *Non-spatiale* et la version *Boîte Urbaine* (*Non-spatial* and *Urban box*).

Résultats et Discussion

L'utilisation de la méthode *Equilibrium Partitioning* pour calculer les facteurs d'écotoxicité dans les sédiments aboutit à quelques inversions notables par rapport à l'écotoxicité aquatique dans le cas des substances hydrophobiques ayant une toxicité aquatique élevée. La résultante faible obtenue en terme d'écotoxicité dans les sédiments pour ce type de contaminants est par conséquence contradictoire par rapport aux objectifs de qualité des sédiments fixés pour le site contaminé. La littérature indique que l'hydrophobicité peut effectivement atténuer l'écotoxicité dans les sédiments, particulièrement pour les HAP, mais il y a aussi des expériences qui indiquent que lorsque l'hydrophobicité augmente, l'ingestion des sédiments devient une voie d'exposition importante, ce qui va à l'encontre des hypothèses de la méthode *Equilibrium Partitioning*. Bien que les facteurs d'effet écotoxicologique pour les sédiments calculés selon la méthode *Equilibrium Partitioning* n'aient pas pu être validés dans la présente étude, il est recommandée ici d'user de précaution lors de leur application et de leur interprétation.

Peu de différences sont observées entre les facteurs de caractérisation pour l'écotoxicité aquatique obtenus à l'aide des versions *Non-spatiale* et *Bassin Versant* du modèle *IMPACT*. Par contre, des différences de un à deux ordres de grandeur existent entre les facteurs de caractérisation produits par les versions *Bassin Versant* et *Bassin Versant Modifié*. Les différences entre les modèles sont plus importantes pour les substances les plus persistantes, ce qui a pour résultat un changement du classement des facteurs de caractérisation, réduisant l'impact relatif pour ce genre de contaminant.

Ces différences entre les modèles sont principalement expliquées par la différence de temps de résidence de l'eau entre la zone d'émission et l'océan, pour lequel aucun impact écotoxique n'est évalué selon le modèle IMPACT. Les versions *Non-spatiale* et *Bassin Versant* ont des temps de résidences similaires, mais la version *Bassin Versant Modifié* a un temps de résidence inférieur de deux ordres de grandeur. Le facteur de devenir d'une substance, et par conséquence l'impact évalué par les différentes versions du modèle, varie directement en fonction du temps de résidence dans le compartiment environnemental concerné. Avec un temps de résidence réduit, la persistance des substances devient moins importante que l'écotoxicité pour l'évaluation des impacts. Cet effet est plus apparent avec les métaux. La plupart des métaux étudiés ici ne sont pas beaucoup plus écotoxiques que les substances organiques, mais ils ne se dégradent pas. Par conséquent, les métaux présentent des diminutions significatives d'impact écotoxicologique lorsque la version *Bassin Versant Modifié* est utilisée.

Peu de différences sont observées entre les facteurs de caractérisation pour la toxicité humaine obtenus à l'aide des versions du modèle *Non-spatiale* et *Bassin Versant.* Par contre, des différences de un à trois ordres de grandeur sont observées entre les facteurs de caractérisation pour la toxicité humaine produits par les versions *Bassin Versant* et *Bassin Versant Modifié.* Ces différences importantes sont attribuable d'une part à une différence de temps de résidence de l'eau entre les différentes versions du modèle et d'autre part au fait que l'eau soit ou non considérée dans le modèle en tant que source d'eau potable (notamment, l'eau de Puget Sound n'est pas utilisée comme eau potable). Des changements de classement entre les facteurs de caractérisation des différentes substances sont aussi observés avec ces différentes versions du modèle, dus à la réduction de l'importance que joue la persistance dans l'évaluation de l'impact toxique d'une substance émise dans des eaux qui atteignent plus rapidement l'océan.

Les facteurs de caractérisation écotoxicologiques pour les émissions dans l'air calculés à l'aide de la version du modèle *Boîte Urbaine* sont 1.92 à 61.7 fois plus élevés que ceux provenant de la version du modèle *Non-spatiale.* Les différences de taux de précipitation et de taux de ruissellement sont identifiées comme les principales causes de ces différences, puisque ils sont responsable du transfert des contaminants de l'air vers le sol et finalement vers l'eau. Le taux de précipitation dans la zone d'émission et dans celles situées sous le vent pour la version du modèle *Boîte Urbaine* est environ deux fois celui du modèle *Non-spatial*, et les taux de ruissellement du modèle *Boîte Urbaine* est environ 5.4 fois plus élevé que celui de la version *Non-spatiale.*

Les facteurs de caractérisation pour la toxicité humaine pour les émissions dans l'air calculés à l'aide de la version du modèle *Boîte Urbaine* sont 1.81 à 702 fois plus élevés que ceux provenant de la version *Non-spatiale.* L'ingestion est la voie d'exposition la plus importante pour la plupart des contaminants, et le fait de tenir compte de la densité de la population avec la boîte urbaine a peu contribué aux différences observées entre les résultats des différentes versions du modèle. Pourtant, les différences de taux de production agricoles, particulièrement en ce qui concerne les produits exposés à l'air, sont identifiées comme des causes importantes de ces différences. La production agricole par unité de superficie dans la zone d'émission et les environs de la version *Boîte Urbaine* est 2.3 fois plus élevée que celle de la version *Non-spatiale.* Cette différence, ajoutée à une déposition au sol plus importante à cause du taux de précipitation élevé a pour résultat que l'ingestion des contaminants via les produits exposés à l'air est 5.5 à 270 fois plus élevée avec la version *Boîte Urbaine.*

L'option d'atténuation naturelle génère des impacts secondaires principalement à cause du diesel utilisé pendant l'échantillonnage et le transport des échantillons, mais ce sont les impacts primaires associés à cette option qui sont les plus importants. Le modèle d'Atténuation Naturelle prédit que le site contaminé va émettre approximativement 290,000 kg de métaux et 3500 kg de contaminants organiques dans l'eau et 8.7 kg de mercure et 480 kg de contaminants organiques dans l'air. L'advection est le processus de disparition des contaminants de la colonne d'eau le plus important. Afin de savoir quel est le principal processus de disparition des différents contaminants présents, les cinétiques de ces processus sont prises en compte. Dans un premier temps la contribution de la dégradation à la disparition totale est estimée, par la suite on regarde quelle est la contribution relative de la re-suspension et l'enfouissement pour expliquer la disparition de ce qui n'a pas été dégradé. La diffusion n'est pas ici un processus important pour le transfert de masse. Après 100 ans, le modèle prévoit l'absence de contaminants dans les sédiments mixtes. La disparition totale des contaminants organiques dans les sédiments profonds au bout de 100 ans est également prédite par le modèle, à l'exception des BPC.

Les impacts sur le site lui-même, associés à la contamination dans les sédiments sur une période de 100 ans, sont estimés à 8.86E+08 PAF-m^3-an, dont 96% causés par le zinc et le cuivre seuls. Les impacts écotoxicologiques hors site pour toutes les émissions à l'eau sur une période de 100 ans totalisent 1.17E+09, 1.55E+09 et 3.66E+06 PAF-m^3-an respectivement pour les modèles *Non-spatial*, *Bassin Versant* et *Bassin Versant Modifié*, dont 96% sont attribués au zinc et au cuivre. Les impacts écotoxicologiques hors site pour l'ensemble des émissions à l'air totalisent 1.91E+05 PAF-m^3-an avec le modèle *Boîte Urbaine* et 1.85E+04 PAF-m^3-an pour le modèle *Non-spatial*, le mercure étant responsable de 99% des impacts dans les deux cas.

Les impacts toxicologiques hors site pour l'ensemble des émissions à l'eau totalisent 6.76E+01, 1.02E+02, et 2.29E-01 DALY pour les modèles *Non-spatial*, *Bassin Versant* et *Bassin Versant Modifié*, respectivement. Les métaux sont responsables de la majorité de l'impact, dont le zinc qui est responsable de 77 à 82% selon le modèle utilisé. Les impacts toxicologiques hors site pour toutes les émissions à l'air totalisent 1.39E-02 DALY pour le modèle *Boîte Urbaine* et 7.09E-03 DALY pour le modèle

Non-spatial, le mercure étant responsable de 89% et 97% de l'impact pour les modèles *Boîte Urbain* et *Non-spatial,* respectivement.

L'ACV des activités de réhabilitation aboutit aux résultats suivants pour les différentes catégories de dommages : 8.59E+02 DALY pour la santé humaine, 1.64E+07 PDF-m^2-an pour la qualité des écosystèmes, 1.07E+09 kg CO_2 eq pour les changements climatiques et 3.51E+10 MJ pour l'utilisation des ressources. La grande majorité de ces impacts est causée par l´utilisation d'un géotextile de polypropylène dans le site d´élimination.

Comme mentionné plus haut, la régionalisation et l'introduction de données spécifiques au site a abouti à l'obtention de résultats significativement différents en termes de facteurs de caractérisation. Cependant, les conclusions qui peuvent ressortir de l´application de ces facteurs de caractérisation sont moins claires et la précaution est recommandée pour l´interprétation des résultats.

Le modèle d'atténuation naturelle et le modèle IMPACT sont sensibles à certains paramètres qui ont une variation naturelle de plusieurs ordres de grandeur, ce qui implique une incertitude significative sur les résultats de ces modèles. La quantification de cette incertitude ne fait pas l'objet de cette étude, cependant comme règle générale, des différences de moins de deux ordres de grandeurs dans les résultats ACV ne peuvent pas être considérées comme significatives.

Quelles que soient la perspective adoptée et la version du modèle IMPACT utilisée, les activités de réhabilitation causent des impacts plus importants que l'atténuation naturelle pour les catégories de dommages « Changements Climatiques » et « Utilisation des Ressources ». Selon la perspective considérant que le site fait partie de l´écosphère, on ne constate pas de différence significative entre les deux options de gestion pour les catégories de dommages « Santé Humaine » et « Qualité des

Écosystèmes », sauf lorsque les facteurs de caractérisation sont calculés avec la version *Bassin Versant Modifié* du modèle IMPACT. Dans ce dernier cas, les impacts de l'atténuation naturelle pour la catégorie « Santé Humaine » deviennent significativement moins importants que ceux des activités de réhabilitation.

Selon la perspective considérant que le site fait partie de la technosphère, le résultat est similaire, sauf que les impacts sur la Santé Humaine et sur la Qualité des Écosystèmes de l'atténuation naturelle sont significativement moindre que ceux des activités de réhabilitation lorsque l'on utilise la version *Bassin Versant Modifié* de IMPACT pour calculer les facteurs de caractérisation.

Les résultats indiquent que l'intégration de données spécifiques au site en ÉICV via la modélisation régionalisée du devenir et de l'exposition peut modifier les conclusions d'une ACV comparant les différentes options de réhabilitation d'un site contaminé. Par ailleurs, une nécessité importante d'améliorer la modélisation de devenir des métaux a été constatée.

Pour l'instant, la capacité des modèles comme IMPACT et le modèle d'atténuation naturelle de décrire le devenir des métaux dépend beaucoup de la validité des coefficients de partition empiriquement dérivés qui ont été utilisés ici. Ces coefficients entraînent des incertitudes significatives dans les résultats, notamment à cause de l'incapacité de tenir compte de la spéciation, qui détermine le devenir et la biodisponibilité des métaux.

Le choix de considérer le site comme faisant partie de la technosphère ou de l'écosphère dépend des objectifs et de la portée de l'ACV. Considérer le site comme faisant partie de la technosphère semble appropriée si l'exposition aux contaminants et limitée. Pour ce projet, le choix d'évaluer le site comme partie de l'écosphère semble justifié, étant donné que l'exposition des organismes benthiques aux contaminants dans les sédiments entraîne des impacts importants.

L'évaluation des impacts primaires en utilisant l'ÉICV permet la comparaison avec les impacts secondaires selon une échelle commune. Les résultats présentés ici ne sont que des indicateurs (et non des mesures), mais il est concevable que le choix entre deux options de gestion soit influencé par l'importance des impacts générés par les activités de réhabilitation par rapport aux impacts générés par le site contaminé. Dans la présente étude, cette comparaison mène à la conclusion que l'atténuation naturelle a moins d'impacts que les activités de réhabilitation. Cette conclusion, qui reste discutable, souligne néanmoins la divergence entre l'analyse de risque et l'analyse du cycle de vie. Cependant, il n'est pas nécessaire que les deux analyses s'accordent, il s'agit de deux outils complémentaires pour l'aide à la décision et cette divergence est seulement le reflet de la capacité de l'ACV à considérer certains impacts dont l'analyse de risque ne tient pas compte.

En dépit des efforts de la présente étude pour améliorer l'évaluation des impacts primaires afin de faciliter leur comparaison aux impacts secondaires, il est probable que les parties prenantes d'un tel projet de réhabilitation utiliseraient de toute façon une pondération subjective pour effectuer une telle comparaison.

En ce qui concerne le site contaminé, il est recommandé que d'autres options de réhabilitation soient recherchées, par exemple de réduire la masse du géotextile, ou de trouver un site d'élimination qui n'a pas besoin d'une telle mesure de sécurité.

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LIST OF ACRONYMS AND ABBREVIATIONS

CBNT	Commencement Bay Nearshore Tideflats
CDF	Confined disposal facility
CIRAIG	Centre interuniversitaire de recherche sur le cycle de vie des produits, procédés et services
ERA	Environmental risk assessment
HUC	Hydrologic unit code
ISO	International Organization for Standardization
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
MNA	Monitored natural attenuation
MWW	Middle Waterway
NRCAN	National Resources Canada
PAF	Potentially affected fraction
PAH	Polycyclic aromatic hydrocarbons
PCB	Polychlorinated biphenyls
PDF	Potentially disappeared fraction
SQO	Sediment quality objective
TSS	Total suspended solids
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WA	Washington State

"Essentially, all models are wrong, but some are useful". (George Box)

1 Introduction

Contaminated sediments have become one of the legacies of modern industrial societies, affecting coastal and surface waters across Canada. Harbour areas and waters bordering urban or industrial areas are frequently recipients of contaminated effluent or accidental spills containing chemicals which will settle in sediments (National Research Council, 1989). A study of more than 21,000 sediment sampling stations across the United States (USEPA, 2002b) classified 26% of the sites as showing probable adverse effects and 49% as showing possible adverse effects. Sediment contamination adversely affects aquatic and marine ecosystems through direct impact on benthic organisms which then continues up through the food chain (Snelgrove *et al.*, 2000; Covich *et al.*, 2004). Studies investigating ecosystem stability have stressed the importance of benthic ecosystem health and biodiversity for the stability and survival of overlying marine and freshwater environments (Snelgrove *et al.*, 1997; Wall *et al.*, 1998; Covich *et al.*, 1999; Palmer *et al.*, 2000; Levin *et al.*, 2001; Austen *et al.*, 2002)

When considering the management of such contaminated sites, an approach is required that ensures a net positive environmental benefit will be gained from any remediation action. Such an approach should be in agreement with the principles of sustainable development, taking into consideration not only the local primary impacts caused by the contamination but also the secondary impacts associated with the remediation activities themselves, which can be distributed far from the site and contribute to regional and global impacts, such as global warming, acidification, eutrophication or smog.

The scale of these secondary impacts for contaminated site remediation projects can be significant. Ellis *et al.* (2008, cited in Ellis & Hadley, 2009) offer the example of a remediation project in New Jersey for which the estimated difference in greenhouse gas emissions between proposed options equalled approximately 2 percent of the annual emissions for the entire state.

Secondary impacts are typically evaluated using life cycle assessment (LCA), which assesses the environmental impacts caused by the sum total of inputs and outputs contributing to the remediation project. These impacts range from those caused by raw material extraction for any products used, to landfill emissions caused by the associated wastes.

The value of LCA in assessing the impacts associated with contaminated site remediation has been demonstrated in previous studies with LCA-based decision-making tools developed for remediation projects within Canada (Diamond *et al.*, 1999), the United States (Vignes, 2001), the Netherlands (Suèr *et al.*, 2004), Germany (Volkwein *et al.*, 1999), Denmark (Suèr *et al.*, 2004) and the United Kingdom (Harbottle, 2008).

Consideration of secondary impacts can have particular relevance for contaminated site management in cases where the appropriate regulatory criteria can not be met. In such situations, a comparison of primary and secondary impacts is particularly useful.

To demonstrate this comparison, the relationship between primary and secondary impacts for many remediation projects is depicted graphically in Figure 1.1. The curve indicates that reduction in risk or primary impact, in this case correlated with the concentration of contaminant, is brought about by an increase in effort, correlated with secondary impacts.

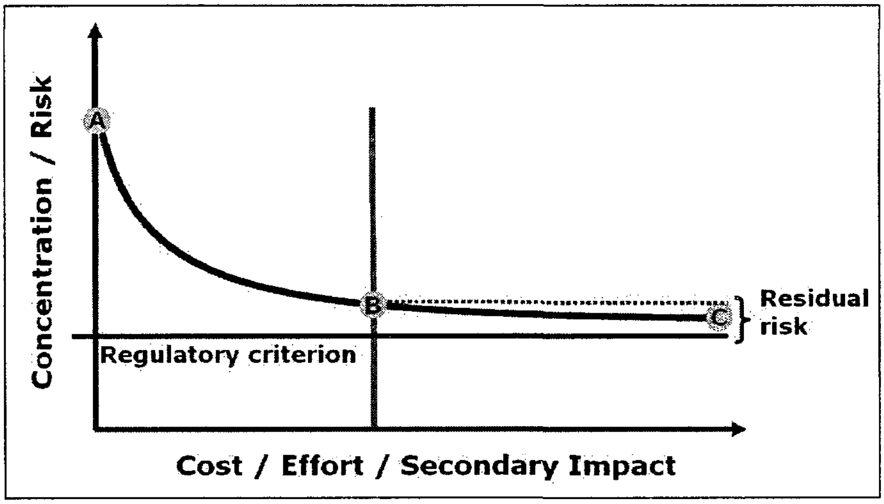


Figure 1.1. Relationship between concentration/risk and remediation effort/secondary impact at a contaminated site for which regulatory compliance can not be achieved. Point A represents the initial contaminated state. Point B represents the risk level achieved by source removal or similar risk mitigation techniques. Point C represents the risk level achieved following increased remediation effort and continued management of residual risk. (After Nicolette, 2006)

For such a situation, it may be possible to significantly reduce the risk with a minimal expenditure of effort (from A to B), perhaps in removing the contamination source, or by putting transport controls into effect. However further reductions in primary impact (from B to C) can not be realized without an increasingly intensive expenditure of effort and generation of secondary impacts. Moreover, in this example, further expenditures do not significantly reduce the risk or result in regulatory compliance. This risk may be caused by residual contamination that is difficult or impossible to extract. Such situations are surprisingly common (National Research Council, 2005, cited in Ellis & Hadley, 2009), making it increasingly worthwhile to evaluate the secondary impacts required to bring about further reduction in primary impacts.

While LCA is useful in evaluating secondary impacts, significant problems exist in its evaluation of primary impacts, due to the generic manner in which life cycle impact assessment (LCIA) methods evaluate toxicity and ecotoxicity. The impact caused by a contaminated site will vary greatly with local conditions, yet this variation is not accounted for by LCIA methods, since all emissions are characterized in the same fashion, regardless of where they occur. Previous studies that measured primary impacts using LCIA methods found that these impacts were very likely to be overestimated (Godin *et al.*, 2004; Dontigny, 2004; Toffoletto *et al.*, 2005; Lesage *et al.*, 2006; Cadotte *et al.*, 2007).

Site-specific data are typically collected for remediation projects as part of the site characterization or environmental risk assessment. The parametrization of LCIA models using these data may allow for a more representative estimation of the primary impacts due to on-site contamination and off-site emissions.

This master's thesis investigates the effect of using spatially differentiated LCIA models to integrate site specificity into the LCA comparing remediation options for a coastal industrial site with contaminated sediments

2 Literature Review

2.1 Contaminated Sediments

Aquatic and marine sediments are loose particles of sand, clay, silt, and other substances that settle at the bottom of a body of water. They come from eroding soil and decomposing organic material and can be carried great distances by wind and water before settling and consolidating (Lick, 2008).

Sediment contamination is a widespread problem found in many industrialized areas (USEPA, 1987, 2002b). Contaminated sediments can result in toxic effects on benthic organisms and throughout the overlying aquatic or marine ecosystem, with contaminants being transferred directly into the water column or travelling through the food chain (Snelgrove *et al.*, 2000; Covich *et al.*, 2004). The most frequently cited pollutants affecting sediment ecosystems in the United States are polychlorinated biphenyls (PCB), pesticides, and polycyclic aromatic hydrocarbons (PAH) (USEPA, 1987, 2002b).

Treatment of contaminated sediments can be difficult and expensive. Remediation challenges include the risk of disturbing intact sediments and polluting the water column, the technical difficulty of handling and storing aqueous material and the possibility that more contaminated sediments underlie the surface material. This latter challenge can complicate the daunting issue of determining a responsible party (USEPA, 1987, 2002b; Kremer *et al.*, 2006).

The assessment of contaminated sediments poses particular difficulties, and weight-of-evidence analyses integrating comparative community assessments with chemical results and laboratory bioassays are frequently required to determine the presence of adverse effects, as any one of these analyses is considered insufficient on its own (Chapman *et al.*, 1987; Burton *et al.*, 2002; Chapman & McDonald, 2005; Chapman, 2007).

2.1.1 Remediation options for contaminated sediments

Numerous options exist for the management of contaminated sediments. Methods can be classified according to one of four management modes (USEPA, 1994):

- Containment in place
- Treatment in place
- Excavation and containment
- Excavation and treatment

The various technology types associated with these four management modes are listed below in Table 2.1.

Table 2.1. Sediment remediation technology types according to management mode (USEPA, 1994).

	In Place	**Excavation**
Containment	• Capping	• Beneficial use • Confined aquatic disposal/capping • Landfill • Confined disposal facility
Treatment	• Bioremediation • Chemical • Immobilization	• Chemical • Biological • Extraction • Immobilization • Physical Separation • Thermal

The present study considers two management options: monitored natural attenuation and dredging with upland disposal in a confined disposal facility.

2.1.2 Monitored natural attenuation

Monitored natural attenuation (MNA) is a contaminated site management option in which contaminant concentrations are reduced to acceptable levels without the assistance of human intervention, using natural processes such as dilution, volatilization or biological and chemical degradation. Within the context of sediment remediation, MNA includes the process of sedimentation and burial, effectively sequestering contaminants below clean surface material (CPEO, 1998; Van Deuren *et al.*, 2002)

The use of MNA typically requires that attenuation processes such as sequestration and degradation rates be evaluated to assure that recovery will occur within an acceptable time frame. Potential human and ecological exposure pathways must also be assessed to ensure that the level of risk is acceptable. MNA must also be accompanied by long-term monitoring to ensure that remediation is occurring as planned (Newell *et al.*, 2002; Van Deuren *et al.*, 2002).

Monitored natural attenuation has the following advantages over other technologies (Van Deuren *et al.*, 2002; Delisle & Greer, 2004):

- Lower cost
- Less waste generation
- Can be applied to all or part of a given site
- Can be used in conjunction with other remediation techniques

In contaminated sediments, natural capping will be one of the dominant attenuation processes. Sedimentation rates will determine the success of such a mechanism for remediating the site (Mulligan & Yong, 2006).

Degradation in sediments will for the most part occur under anoxic conditions. However mixing action and irrigation due to bioturbation can increase the exposure of contaminants to oxic conditions, which can stimulate aerobic degradation via biological or chemical means (Banta & Anderson, 2003; Mulligan & Yong, 2006). The presence of oxic or anoxic conditions will also determine redox conditions for metals, affecting their mobility as well as their availability as electron acceptors for organic matter decomposition and other oxidation reactions (Banta & Anderson, 2003; Mulligan & Yong, 2006). The activity of benthic infauna can increase degradation through their own metabolism of contaminants, but may also complicate remediation efforts by releasing sequestered chemicals to the water column (Granberg *et al.*, 2008).

A thorough understanding of these mechanisms is required if the feasibility of MNA as a management option is to be demonstrated.

2.1.3 Dredging and Disposal in a Confined Disposal Facility

Dredging is a mechanical removal treatment in which the bed of the waterway is penetrated, raked, cut or scoured to dislodge sediment. The collection of the sediment is then achieved either mechanically, such as with a clamshell bucket, or hydraulically through a pipe (USEPA, 1994a).

Once collected, various options exist for sediment disposal, one of the most frequently used being the employment of confined disposal facilities (CDFs) (USEPA, 1994a; Miller, 1998). CDFs usually consist of earthen dikes designed to function as settling basins so that dredged material placed into the CDF, either mechanically or hydraulically, will settle out of suspension before the supernatant water is discharged at the other end of the facility (USEPA, 1994a). A simplified schematic a confined upland sediment disposal facility is given in Figure 2.1. The outflow rate is usually controlled by a weir and water quality monitoring is conducted near the discharge to ensure that the facility is not a source of contamination. To this end, the Great Lakes Commission (2003; cited in Olsta, 2003) has published results which indicate the effectiveness of CDFs in preventing external contamination. Prevention of groundwater contamination through the bottom of the CDF can be achieved with the use of a geotextile liner. At the end of a CDF's useful life, sediments can be compacted, the site graded for positive drainage and a cap or cover put over top of the dredged material (USEPA, 1994a; Olsta, 2003).

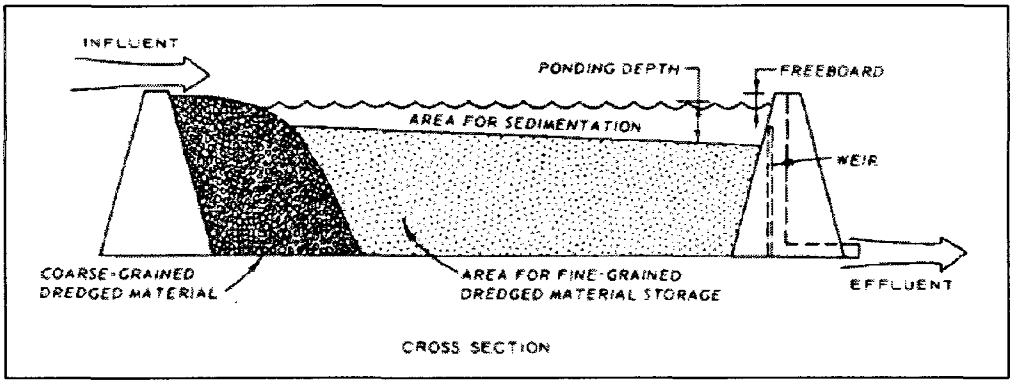


Figure 2.1. Simplified schematic of a confined upland sediment disposal facility (USEPA, 1994a).

2.2 LCA

Life cycle assessment was developed in order to evaluate the environmental performances of different products or services, using a holistic approach that considers all phases in the life cycle of the system (product or service) in question. This evaluation can include material extraction, manufacturing and transportation as well as the associated "end-of-life", looking at the final disposal and waste products associated with the system. This assessment considers all material and energy inputs and outputs throughout the system's life cycle and evaluates their potential contribution to environmental impacts occurring at both local and global scales. Impacts typically evaluated include (Udo de Haes *et al.*, 1999):

- Global warming potential
- Ozone depletion
- Eutrophication
- Acidification
- Human toxicity
- Ecotoxicity
- Photochemical ozone potential (smog)

- Resource depletion
- Land use

The International Standards Association has created a standard for LCA methodology, ISO 14040 (2006a) in which four phases of LCA are described: (1) Goal and scope definition, (2) Inventory analysis, (3) Impact assessment, and (4) Interpretation. Figure 2.2 depicts the four phases of LCA. The two-way arrows between the phases indicate the iterative nature of LCA, in which each phase can be revised after interpretation, ensuring that the appropriate processes, level of detail and impact assessment method are being considered.

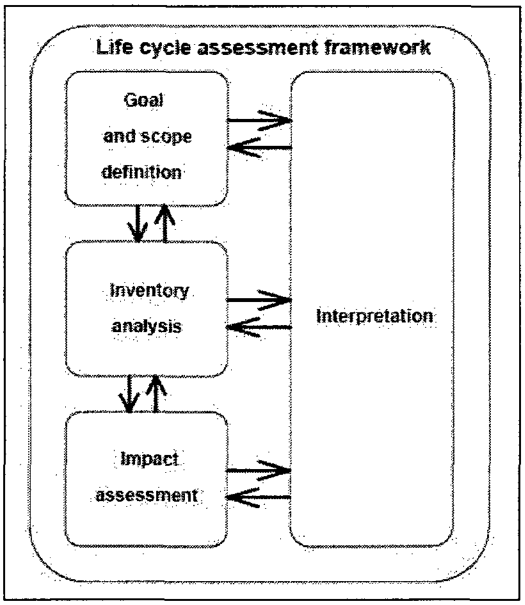


Figure 2.2. Life cycle assessment framework (ISO, 2006a)

2.2.1 Goal and Scope Definition

In this phase, the expected application of the assessment, the reason for its being carried out and the intended audience are defined (ISO, 2006a). This phase also is used to define the function, functional unit and reference flow associated with the system being studied. As LCA is typically used for comparative assessments, each system being evaluated must fulfill equivalent functions. The functional unit quantifies the means by which this function will be carried out and the reference flow indicates the amount of product or service that will be required to produce the functional unit. The system of processes that comprise the product or service being analyzed is also defined during this phase, which serves to set boundaries for the assessment.

2.2.2 Inventory Analysis

The most labour-intensive stage of a LCA is often the inventory analysis, during which all material and energy inputs that flow through the product system are accounted for, as well as all outputs, such as waste or emissions. The life cycle inventory (LCI) is described by process flow diagrams, such as the one shown in Figure 2.3. At its most detailed level, such diagrams break down the system being studied into unit processes belonging to the man-made environment, or *technosphere* and their interaction with the natural environment, which for modeling purposes can be designated the *ecosphere*. Unit processes are made up of elementary and intermediate flows. Elementary flows represent inputs from the ecosphere, such as energy and raw materials and outputs such as water, air and soil emissions. An intermediate flow consists of a product, material or energy that is transferred within the technosphere between unit processes (see Figure 2.4).

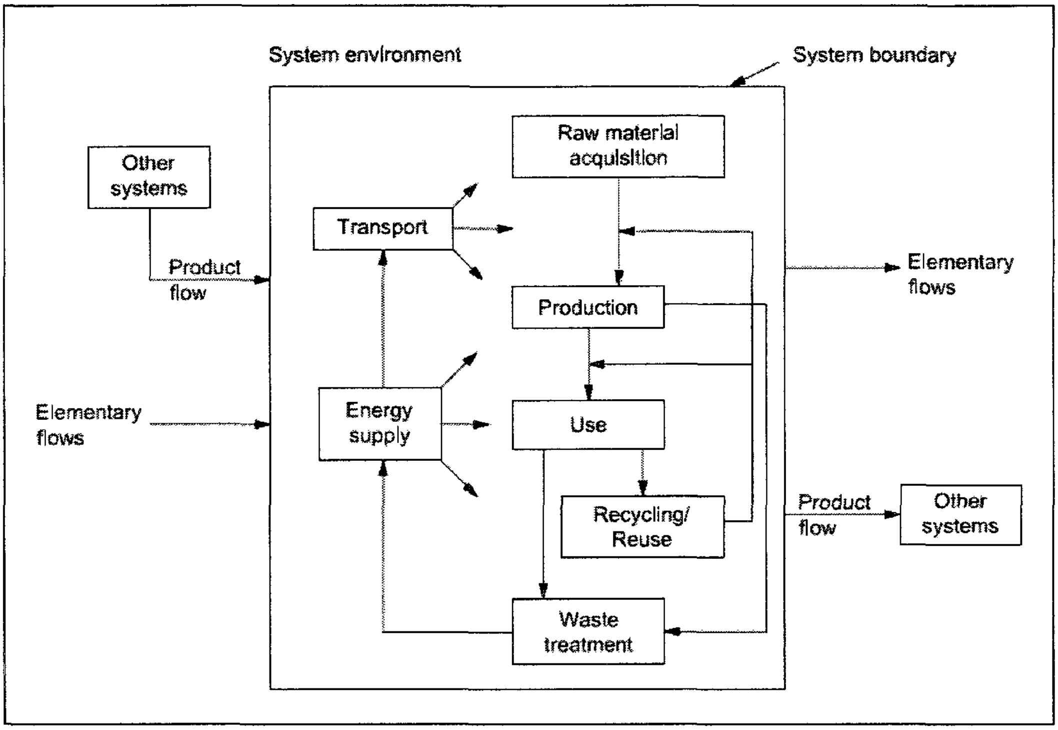


Figure 2.3. Example of a LCA product system (ISO, 2006a).

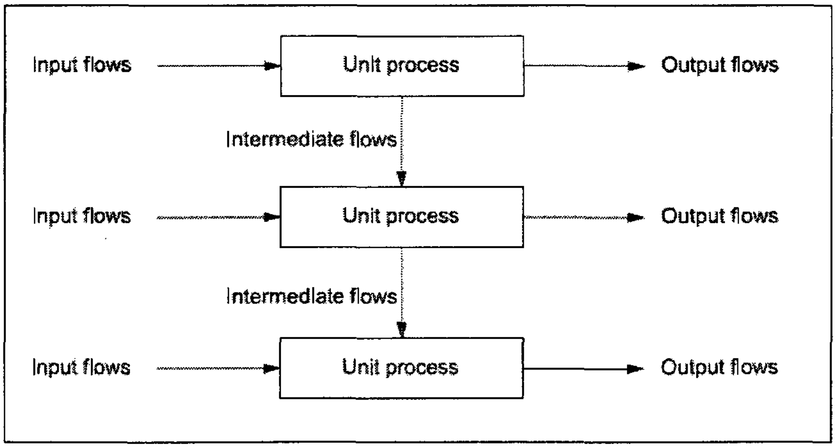


Figure 2.4. Example of a set of unit processes within a product system (ISO, 2006a).

The total quantity of inputs and outputs making up the LCI for the system being studied is calculated as a function of the reference flow defined in the goal and scope, with elementary and intermediate flows being normalized to this quantity.

Data to build the LCI may come from a variety of sources, with primary (original) data regarding unit processes being considered of the highest value. These data are typically not available for all processes and data are often taken from literature sources and databases, such as Ecoinvent (Frischknecht *et al.*, 2005).

2.2.3 LCIA

The impact assessment phase determines the environmental impacts associated with the inputs and outputs quantified during the inventory analysis, with the results typically being compiled according to impact category (listed above). According to the ISO standard 14040 (ISO, 2006a), obligatory elements of the impact assessment phase include (i) the selection of impact categories, category indicators and characterization models, (ii) the assignment of LCI results (classification), and (iii) calculation of category indicator results (characterization). This can be followed by the optional elements normalization, grouping and weighting of the results.

Conversion of the LCI results into impact categories is done via characterization factors that describe the impact that an inventory quantity will have in terms of the common unit used by the impact category. For example, an acidification impact category may characterize all acidifying inventory items in terms of SO_2 equivalents released to the air. The calculation of the impact caused by acidifying substances would then be:

$$\mathrm{Impact} = \sum_{i} \mathrm{CF}_i \cdot e_i \qquad \text{(Eq. 2.1)}$$

where:

e_i = emission mass for substance *i* [kg]

CF_i = characterization factor for substance *i* [$\mathrm{kg_{eq}}$ SO_2 /kg substance *i* emitted]

Impact assessment indicators can evaluate impacts at their midpoint or at their endpoint, according to where in the cause-effect chain the impact is measured. Indicators that assess at midpoint, give results in terms of quantities that contribute to a given environmental impact (e.g., kg_{eq} chloroethylene in air for human toxicity), whereas those that assess at endpoint give results in terms of damages caused by an environmental impact (e.g., Disability Adjusted Life Years, DALY for human health impacts).

2.2.4 IMPACT 2002+

The life cycle impact assessment method used for this project is IMPACT 2002+ (Jolliet *et al.*, 2003). This method provides characterization factors for 15 midpoint impact categories, which can also be summarized with 4 endpoint impact categories, as presented in Table 2.2.

Table 2.2. IMPACT 2002+ midpoint and endpoint indicators with units (Jolliet *et al.*, 2003). The midpoint indicators are grouped according to the endpoint indicators to which they contribute.

Endpoint category	**Midpoint category**
Human health [DALY]†	Toxicity – Carcinogens [kg_{eq} chloroethylene in air] Toxicity – Non-carcinogens [kg_{eq} chloroethylene in air] Respiratory inorganics [kg_{eq} PM2.5 in air] Respiratory organics [kg_{eq} ethylene in air] Ionizing radiation [Be_{eq} C^{14} in air] Ozone layer depletion [kg_{eq} CFC-11 in air]
Ecosystem quality [PDF-m^2-year] ‡	Aquatic ecotoxicity [kg_{eq} triethylene glycol in water] Terrestrial ecotoxicity [kg_{eq} triethylene glycol in water] Terrestrial acidification/nutrification [kg_{eq} SO_2 in air] Land occupation [m^2_{eq} organic arable land-year] Aquatic acidification [kg_{eq} SO_2 in air] Aquatic eutrophication [kg_{eq} PO_4^{3-} in water]
Climate change [kg CO_2 eq]	Global warming [kg CO_2 eq]
Resources [MJ]	Non-renewable energy [MJ total primary non-renewable] Mineral extraction [MJ additional energy]

†: Disability Adjusted Life Years

‡: PDF = Potentially Disappeared Fraction

2.2.4.1 Calculation of Toxicity and Ecotoxicity impacts within IMPACT 2002+

Of particular interest for this project are the human toxicity and aquatic ecotoxicty impact categories because their significance for contaminated site management projects. In IMPACT 2002+, ecotoxicity characterization factors (CF) are comprised of a fate factor (FF), describing the environmental fate of the substance in question, and an effect factor (EF), which describes the effect that the substance will have on flora and fauna. The calculation of characterization factors from fate factors and effect factors is shown below in Equation 2.2:

$$CF = FF \bullet EF \qquad \text{(Eq. 2.2)}$$

Human toxicity characterization factors are comprised of intake fractions (iF) and effect factors (EF) as shown below in Equation 2.3:

$$CF = iF \bullet EF \qquad \text{(Eq. 2.3)}$$

The characterization method reflects the cause-effect chain from chemical emission to ecological damage. The framework for the ecotoxicity as well as human toxicity characterization methods are shown below in Figure 2.5.

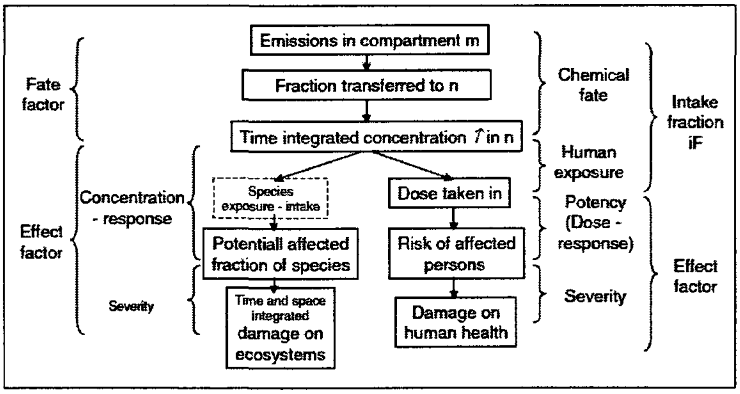


Figure 2.5. Methodological framework for calculating ecotoxicity and human toxicity characterization factors (Jolliet *et al.*, 2003).

2.2.4.2 Fate factors and intake fractions

Fate factors and intake fractions are determined using the IMPACT multimedia fate and exposure model which estimates the distribution of a substance emitted into one environmental compartment (air, soil, water) that will be transferred into another, as well as the fraction of this emission to which people will be exposed either through ingestion or inhalation.

IMPACT is a multimedia chemical fate and exposure model, based on mass balances linking numerous environmental compartments including: soil, freshwater, air, vegetation, sediments and the sea (Humbert *et al.*, 2009). A schematic framework of the model showing the various compartments and mass transfers is provided in Figure 2.6.

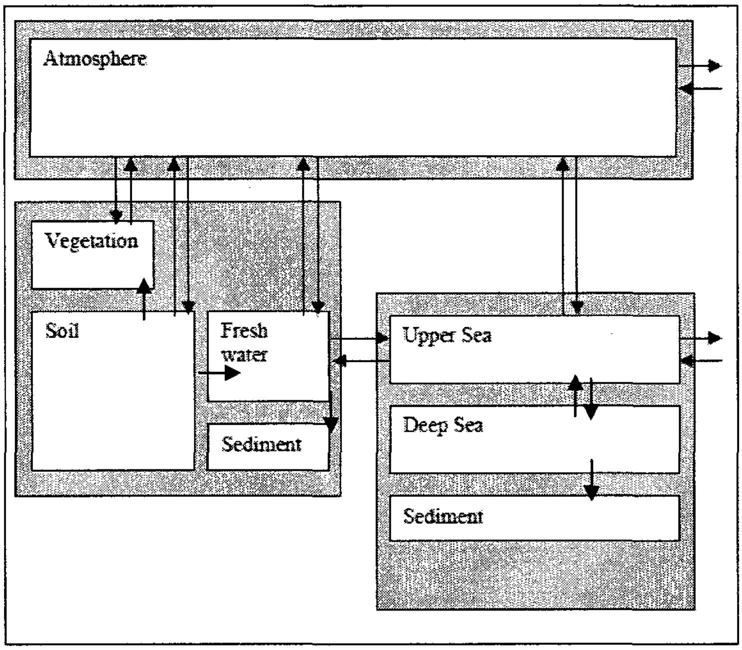


Figure 2.6. Schematic framework of the IMPACT multimedia fate and exposure model showing modeled environmental compartments and mass transfers (Pennington *et al.*, 2005).

The model assumes a steady state to solve the mass balances and predict changes in environmental concentrations and the resultant exposure to humans and environmental receptors due to emissions occurring in either soil, air or freshwater. Human intake is calculated via inhalation and ingestion. Ingestion pathways include drinking water, incidental soil consumption, agricultural products and animal products. Agricultural products include exposed (above-ground) and unexposed (below-ground) produce. Animal products include beef, pig, poultry, eggs, fish, and milk. The intake fraction can then be calculated using agricultural production levels, the population served by a water source and the population distribution for inhalation exposure routes (Pennington *et al.*, 2005; Humbert *et al*, 2009). For the purpose of calculating ecotoxicity fate factors, a default 100% exposure is assumed for environmental receptors existing in soil or freshwater.

Two non-spatial versions of IMPACT have been developed, an original Western Europe model (Pennington *et al.*, 2005) and a Canada model (Manneh *et al.*, 2009), each representing the respective geographic regions as a single box within a larger world box. These models have a single air layer and one coastal zone. Advective flow via water and air occur between the modeled region and the world box, which is non-spatially differentiated.

Spatially differentiated versions of the IMPACT model have also been developed. IMPACT North America is a spatially differentiated version of the IMPACT 2002 model which covers spatial scales including indoor, urban and regional environments. The environmental compartments, such as those indicated in Figure 2.6, are differentiated geographically (where appropriate) according to delineations described below in order to reflect environmental conditions and human exposure for the North American context. Environmental parameters such as the dimensions of water, soil and biomass compartments, meteorological conditions and hydrological conditions are determined for each spatially differentiated zone. Human exposure parameters such as population data, drinking water and food production are also specified. A list of parameters that can be adapted to a specific geographic region is provided in Table A1.1 of Appendix 1.

The regional environment for soil and water compartments is differentiated according to NRCAN (2003) and USGS (Seaber *et al.*, 1987) hydrologic unit delineations, resulting in 523 watersheds, with average advective flow calculated between them. Air zones for the North American model are made up of 831 cells, each measuring 2° latitude by 2.5° longitude. A mixing height of 1000 m is set for the air layer, with advective flow calculated using average wind speeds from the GEOS-Chem model (Jacob & Yantosca, 2005). North America is surrounded by five coastal zones (Pacific, Atlantic, Arctic, Bering, and Hudson Bay) each linked to the world box ocean (NGDC, 2006). Figure A1.1 in Appendix 1 presents the various watersheds, airsheds and coastal zones modeled by IMPACT North America.

An urban environment is included in the IMPACT North America model in order to estimate impacts due to air emissions occurring in areas with highly variable population density. The watershed and airshed within which the urban box is nested must be specified for the model.

2.2.4.3 Ecotoxicity effect factors

Ecotoxicity effect factors are calculated using the Potentially Affected Fraction (PAF) indicator. For a given environmental concentration, the PAF is the fraction of species that will be exposed above a given toxicity endpoint. The PAF can be taken directly from a Species Sensitivity Distribution (SSD), which represents the cumulative distribution of toxicity endpoints, such as the NOEC (No Observable Effect Concentration), EC50 (50% effect concentration) or LC50 (50% lethal concentration), for a collection of species (Payet, 2004).

IMPACT 2002+ calculates aquatic ecotoxicity effect factors according to the Assessment of Mean Impact (AMI) method, which uses the EC50 as its toxicity endpoint for constructing the SSD. For a given chemical emission, the effect factor is taken from the average gradient (dPAF/dC) of the SSD, using a linear interpolation between the origin and the 50% hazard concentration, the $HC50_{EC50}$. The $HC50_{EC50}$ represents the geometric mean of the EC50 values (the 50th percentile concentration) for that chemical. Figure 2.7 shows a simplified SSD of EC50 concentrations and the calculated HC50 concentration.

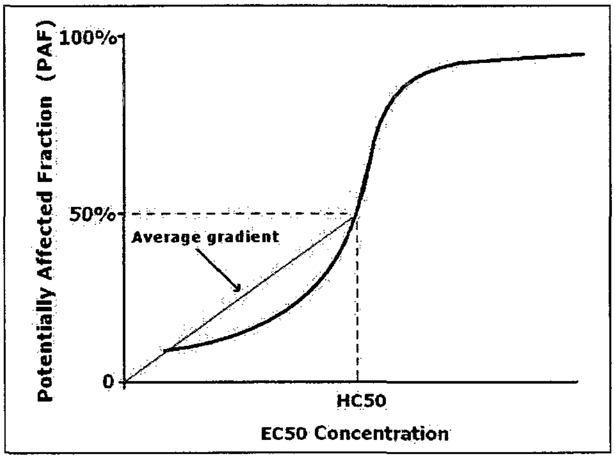


Figure 2.7. Simplified species sensitivity distribution (SSD), with 50% hazard concentration (HC50) and average gradient (dPAF/dC) shown, after USEPA (1998).

The calculation of the effect factor (**EF**) can be simplified then to Equation 2.4 (Payet, 2004):

$$\mathrm{EF} = \mathrm{d}\frac{\mathrm{PAF}}{\mathrm{dC}} = \frac{0.5}{\mathrm{HC50}} \qquad \text{(Eq. 2.4)}$$

For a given chemical emission, the fate factor determines the increase in concentration within the relevant environmental compartment. This concentration increase is then multiplied by the volume affected (m^3) and by the duration of the impact (year). After multiplication by the effect factor, the ecotoxicological impact has units of PAF-m^3-year.

2.2.4.4 Human toxicity effect factors

Effect factors for human toxicity impacts are calculated according to the formula shown in Equation 2.5 (Jolliet *et al.*, 2003):

$$\mathrm{EF} = \beta \cdot \mathrm{D} \qquad \text{(Eq. 2.5)}$$

where:

β = dose-response slope factor [risk of incidence/kg intake]

D = severity [DALY/incidence]

The toxicological measure used for determining the dose-response slope factor is the effective dose inducing a 10% response over background levels, the ED_{10}, as per the calculation show in Equation 2.6:

$$\beta = \frac{0.1}{\mathrm{ED}_{10}} \cdot \frac{1}{\mathrm{BW} \cdot \mathrm{LT} \cdot \mathrm{N}_{365}} \qquad \text{(Eq. 2.6)}$$

where:

ED_{10} = effective dose inducing a 10% response over background level [mg/kg/day]

BW = average body weight [kg]

LT = average lifetime [year]

N_{365} = number of days per year [days/year]

Preliminary slope factors were calculated from animal bioassays, making extrapolations from TD50 (50% toxic dose), No Observed Effect Level and Lowest Observed Effect Level data (Jolliet *et al.*, 2003). Default severity values were 13 and 1.3 DALY for incidences of carcinogenic and non-carcinogenic effects, respectively.

2.2.4.5 Characterization of metals

Multimedia models such as IMPACT rely heavily on partitioning coefficients and parameters describing bioavailability (e.g., bioconcentration factors) in order to predict contaminant fate and exposure. It should be noted that these models are poorly suited to evaluate the environmental fate of metals, as speciation is not accounted for. Speciation can affect the fate, bioavailability and toxicity of metals (Gunn *et al.*, 1989; Petronio *et al.*, 2000; UNEP, 2004; Slaveykova, 2009).

Currently, empirically-derived partitioning coefficients for metals are used in order to predict the fate and exposure due to metal emissions (USEPA, 2007). The validity of such partitioning coefficients is unknown, however such parameters add significant uncertainty to modeling results (UNEP, 2004; Haye *et al.*, 2007). As a result, toxicity and ecotoxicity characterization factors developed for metals must be interpreted with caution.

2.3 Application of LCA to Contaminated Site Remediation

In the interest of aligning remediation projects with the principles of sustainable development, site remediation practitioners must take a holistic approach when evaluating contaminated sites and the options available to manage them. The use of LCA allows for management options to be evaluated based not only on the primary impacts caused by the contamination, but also based on the secondary impacts caused by the remediation activities themselves (Lesage *et al.*, 2007; Cadotte *et al.*, 2007, Suèr *et al.*, 2004).

LCA has already been demonstrated as a useful decision-making tool for contaminated site remediation, with LCA-based decision-making tools having been developed for remediation projects within Canada (Diamond *et al.*, 1999), the United States (Vignes, 2001), the Netherlands (Suèr *et al.*, 2004), Germany (Volkwein *et al.*, 1999), Denmark (Suèr *et al.*, 2004) and the United Kingdom (Harbottle, 2008).

LCAs have taken a variety of approaches for assessing impacts associated with site remediation projects. While secondary impacts are always assessed, primary impacts due to on-site contamination may or may not be. As a further qualification regarding the perspectives taken in the literature, the study boundary may include the contaminated site as part of the ecosphere, in which case on-site contamination is evaluated using LCIA methods. Alternatively, LCA may only consider primary impacts due to emissions from the site, ignoring on-site impacts. This approach effectively considers the site as part of the technosphere. A similar approach is used in LCA when considering the application of pesticides to agricultural fields, with the impact of pesticide emissions from the field (e.g., from runoff) being assessed but impacts to the field being ignored (Hellweg & Geisler, 2003).

This section of the literature review considers various approaches that LCA studies have taken in evaluating contaminated site remediation projects. For the present study, consideration of the contaminated site as part of the ecosphere will be referred to as the "ecosphere perspective" and similarly, consideration of the contaminated site as part of the technosphere will be referred to as the "technosphere perspective".

2.3.1 Primary impacts not evaluated

In certain cases, LCA studies of site remediation projects simply do not evaluate primary impacts. This can be achieved by assuming that all remediation options will have the same performance with respect to contaminant removal. Blanc *et al.* (2004) and Bayer & Finkel (2006) took this route for their LCA assuming an equivalency amongst remediation options. A consideration of only secondary impacts effectively assumes a technosphere perspective.

2.3.2 Separate impact categories within LCIA

Some site remediation LCA studies have evaluated primary impacts using a separate set of indicators. Diamond *et al.* (1999) and Page *et al.* (1999) performed some of the first site remediation LCA studies in North America. Owing to the generic nature of LCIA, the authors suggested having two different sets of metrics for primary and secondary impacts, (Diamond *et al.*, 1999). An ecosphere perspective was taken, with site-related (primary) impacts described using land use metrics and the calculation of a residual human toxicity burden. The residual human toxicity burden (RTB) was calculated as the ratio of an appropriate toxicity-based reference concentration (e.g., reference dose) to the mean concentration remaining in the soil above background levels (Page *et al.*, 1999).

In a similar approach, Beinat *et al.* (1997), Volkwein *et al.* (1999) and Harbottle *et al.* (2007, 2008) used standardized exposure models to estimate risk associated with on-site contamination. Separate indicators for primary impact were created by comparing the resulting exposure measures to toxicity and ecotoxicity benchmarks. The site was treated as part of the ecosphere or technosphere depending on the possibility for exposure to on-site contaminants.

2.3.3 Primary impacts assessed within LCIA

In several cases, primary impacts from contaminated sites have been assessed within the framework of LCIA. Several LCA research projects to assess the impacts associated with various aspects of site remediation have been carried out at the CIRAIG (Centre interuniversitaire de recherche sur le cycle de vie des produits, procédés et services) at the École Polytechnique de Montréal. In all studies, the contamination was counted as part of the LCI and evaluated as part of the LCIA, using various LCIA methods to evaluate the primary impacts. The CIRAIG studies and the LCIA methods used are listed below in Table 2.3.

Table 2.3. Contaminated site remediation LCA projects conducted at the CIRAIG (Centre interuniversitaire de recherche sur le cycle de vie des produits, procédés et services), École Polytechnique de Montréal

Author (year)	Object of Study	LCIA Method
Toffoletto *et al.* (2005)	Ex-situ bioremediation of diesel-contaminated soil	EDIP 97
Godin *et al.* (2004)	Remediation of spent pot lining landfill	IMPACT 2002+
Dontigny (2004)	City of Montreal management of contaminated soil	EDIP 97
Cadotte *et al.* (2007)	Remediation of diesel-contaminated site	TRACI
Lesage *et al.* (2006, 2007)	Brownfield management options	IMPACT 2002+

Both Toffoletto *et al.* (2004) and Dontigny (2004) took ecosphere perspectives in studies evaluating a diesel-contaminated site and the City of Montreal's management of contaminated soil, respectively. In each of these studies, exposure to the contaminated soil was limited and it was noted that the primary ecotoxicity impacts were almost certainly overestimated.

Cadotte *et al.* (2007) evaluated the primary and secondary impacts associated with the remediation of a petroleum hydrocarbon-contaminated site with an LNAPL plume moving towards a river. The residual contaminant concentrations remaining in the soil following treatment were used to evaluate the primary impacts, with aquatic ecotoxicity characterization factors being used for the evaluation. The assumption used here that all contamination would eventually enter the water likely resulted in an overestimation of the primary impacts.

Lesage *et al.* (2006, 2007) used LCA to study brownfield management options. An ecosphere perspective was taken and residual soil contamination was entered directly into the LCI as "emissions to soil". Soil contamination impacts were calculated for all options, including the option in which the contamination was left in place, but the risk was mitigated by minimizing exposure. Taking the ecosphere perspective likely resulted once again in an overestimation of primary impacts.

Godin *et al.* (2004) performed a site-specific groundwater transport simulation to estimate the quantities of contaminants leaving the site in a LCA study of a spent pot-lining landfill. A technosphere perspective was taken with impacts only being evaluated once contaminants left the site and entered the water.

The work by Godin *et al.* (2004) represents the highest level of detail of site-specific data integrated into a site remediation LCA. Contaminant transport modeling was supported by several sampling campaigns that yielded measurements of chemical concentrations as well as on-site hydrogeological and geochemical properties.

While these efforts allowed for a better estimate of contaminants leaving the site, the characterization of the primary impact in this study still remains about as general as those found with other contaminated site LCAs. The multimedia-fate model parametrized for average Canadian conditions did not necessarily improve the prediction of the fate of these contaminants in the local environment or the exposure to which local organisms would be subjected. Godin *et al.* (2004) concluded that while the results of the LCIA showed that the leachate from the landfill dominated the ecotoxicity impact and counted for a considerable amount of the total impact for each management option, an appropriate decision regarding these options could not be made without the use of an ecological risk assessment.

LCA has been successful to a certain degree in evaluating contamination site remediation projects, particularly in assessing secondary impacts and identifying "hotspots" that can make certain options unfavourable. However, the relevance of LCA for contaminated sites management is limited by the generic manner in which LCIA treats primary impacts. The impact caused by localized contamination will vary greatly with the local conditions, yet this variation is not represented with LCIA methods, since all emissions are characterized in the same fashion, regardless of where they are emitted.

The characterization factors used in LCIA are meant to model fate and effects in a manner that can be applied broadly, given that the precise location and affected ecosystem for most life cycle inventory emissions will not be known. Contaminant transport and partitioning on a local scale will usually be poorly modeled by LCIA fate factors. The relative size and importance of environmental compartments and transport mechanisms will differ in the local environment from those used to calculate fate factors. Effect factors that estimate an average response to a contaminant may not represent the local ecosystem response. Effect factors are typically based on the average response of a wide range of organisms, many of which will not be found in the local environment. Moreover, an assumption of 100% exposure to all contaminants is assumed for organisms dwelling within a given environmental compartment. Consequently, when primary impacts are assessed with LCIA methods, the result is typically a greatly overestimated ecotoxicity impact (Godin *et al.*, 2004; Dontigny, 2004; Toffoletto *et al.*, 2005; Lesage *et al.*, 2006; Cadotte *et al.*, 2007).

The ability of LCA to evaluate the primary impacts associated with contaminated sites appears dependent on the correct application of LCIA methods in assessing the on-site contamination and the degree to which these methods can represent the local environment. The correct application of LCIA methods is reflected in the perspective taken with respect to the contaminated site. Improving the ability of LCIA methods to represent local conditions can be achieved through efforts to integrate site-specificity into LCIA models.

2.4 Site-specificity and spatial differentiation in LCA

Efforts to integrate site-specificity into LCA have primarily been carried out via spatial differentiation of LCIA methods. The integration of spatial differentiation in LCIA has been advocated by numerous authors (Huijbregts *et al.*, 2000; Sonnemann *et al.*, 2003; Bellekom *et al.*, 2006; Potting & Hauschild, 2006) citing the difference in results that spatial differentiation can produce.

Much work has been done investigating the difference in characterization factors that results from introducing spatial differentiation to multimedia fate and exposure models (MacLeod *et al.*, 2001; Pennington *et al.*, 2005; Rochat *et al.*, 2006; Humbert *et al.*, 2009; Manneh *et al.*, 2009). Most recently, Manneh *et al.* (2009) found that spatially differentiating the IMPACT model for Canada into sub-watershed zones resulted in a higher variability in intake fractions for organic contaminant emissions than when the model used a spatial resolution of ecozones or provinces. Humbert *et al.* (2009) evaluated the differences that result from introducing urban population density to evaluate air emissions, finding that intake fractions compared to rural regions were an order of magnitude higher. In applying a spatially differentiated IMPACT model for North America to the life cycle of diesel fuel, it was found that intake fractions could vary by up to six orders of magnitude, depending on population density and agricultural output.

Sonnemann *et al.* (2003) and Nigge (2000) describe methods of increasing the level of site-specificity in fate or impact models for highly polluting processes. These methods, used to assess the risks posed by stack emissions, create different classes of sites, characterized by parameters such as population density, wind speed and temperature.

Similarly Nishioka *et al.* (2006; see also Nishioka *et al.*, 2002) integrated site-dependency into an evaluation of insulation standards for new homes by creating different classes of power-plant emission scenarios using dispersion modelling, concentration-response functions and epidemiological data that varied depending on the region studied.

2.5 Outlook for integration of site-specific data into LCIA for evaluation of primary impacts

Two options exist for integrating site-specific data into toxicity and ecotoxicity characterization factors: via the fate factor or via the effect factor. For the purpose of evaluating primary impacts associated with contaminated sites, the integration of site-specific data to assess local effect faces significant difficulties. Theoretically, bioassay results from an environmental risk assessment (ERA) could produce a series of EC50 concentrations such that a local HC50 could be determined, thereby allowing the calculation of PAF values (Payet, 2004). This value would be multiplied by the affected volume and the time frame considered giving commensurate units for ecotoxicity of PAF-m^3-year. Several obstacles exist to this approach however. Typically, bioassays for ERA are only performed on a few key species and can not be considered a comprehensive measure of effect (USEPA, 1998). Moreover, bioassays are increasingly being used in concert with field studies and comparative community analyses as part of weight-of-evidence approaches to ERA (Burton *et al.* 2002; Chapman & McDonald, 2005). These approaches are required because the cause-effect relationships implied by bioassays, and needed for the development of LCIA effect factors frequently do not exist. That is, while bioassay results might indicate that a toxic effect exists, community analyses might indicate that there is none.

The interpretation of bioassay results faces certain difficulties as well. Bioassays could theoretically provide PAF values for discrete samples across a contaminated site as indicated above. As contaminated sites are rarely polluted by single chemicals, these PAF values would indicate the impact associated with the mix of chemicals found in the bioassay sample (e.g., $PAF_{Sample1}$). It would be rare that single-substance effect factors (with units PAF/kg substance) could be developed without making dubious assumptions regarding mixture toxicity.

Given the difficulties facing the creation of local effect factors for contaminated sites and the success of previous authors in integrating spatial differentiation into multimedia fate models, the most promising route for improving primary impact assessment through increased site-specificity appears to be through improved fate modeling.

3 Hypothesis and Objectives

3.1 Hypothesis

The hypothesis to be tested with the present study is the following:

The conclusions of a life cycle assessment to evaluate contaminated site remediation options can be changed according to the level of spatial differentiation integrated within the life cycle impact assessment method.

3.2 Objectives

1. Develop a means of evaluating the life cycle inventory of monitored natural attenuation as a management option for a coastal site with contaminated sediments so as to account for the on-site contamination and off-site water and air emissions.

2. Integrate site-specific information into a life cycle impact assessment method, so as to better represent the environment surrounding a contaminated site and evaluate off-site water and air emissions.

3. Compare two contaminated site management options using life cycle impact assessment, the first with significant primary impacts and little secondary impacts, the second with little primary impacts and significant secondary impacts.

4 Methodology

This section presents the study methodology and a description of the case study LCA. As part of the LCI, a mass-balance model for determining contaminant concentrations in sediment and the overlying water column is presented. The LCIA method is described, introducing the sediment ecotoxicity effect factors used to assess on-site impacts. Modifications to the LCIA method also include changes made to the fate models used to calculate toxicity and ecotoxicity intake fractions and fate factors.

4.1 Case Study LCA

The case study for the project is taken from the Middle Waterway ("MWW" hereafter) contaminated site remediation project, one of nine problem areas that make up the Commencement Bay Near Shore/Tide Flats (CBNT) Superfund site (USEPA, 1989). The study considers the remediation of a section of MWW and uses data taken from the site investigation and environmental risk assessment (Anchor & Foster Wheeler, 2001), but as data availability was limited, certain assumptions have been made regarding the site. As such, the remediation project presented here should be considered a hypothetical case study for academic purposes only.

4.1.1 Site description

The site is located in Tacoma, Washington, at the southern end of the main basin of Puget Sound. Figure 4.1 indicates the location of MWW and Commencement Bay in

Puget Sound. The problem areas associated with the CBNT Superfund site are shown in Figure 4.2.

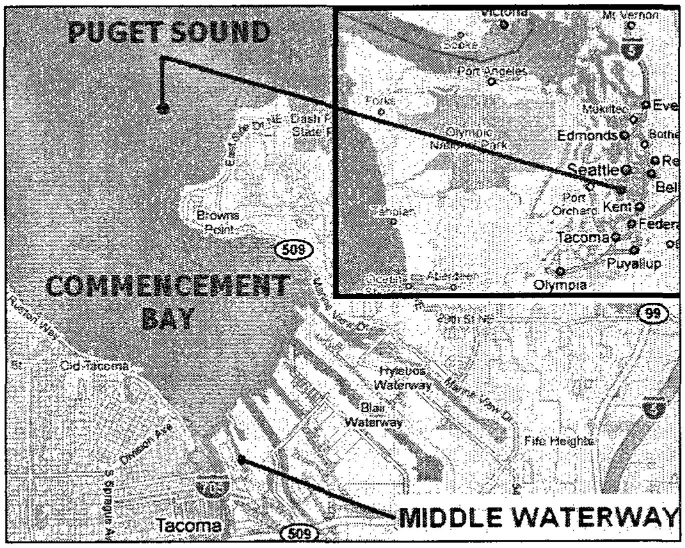


Figure 4.1. Location of MWW and Commencement Bay in Puget Sound, Washington, USA (Google Maps, 2009).

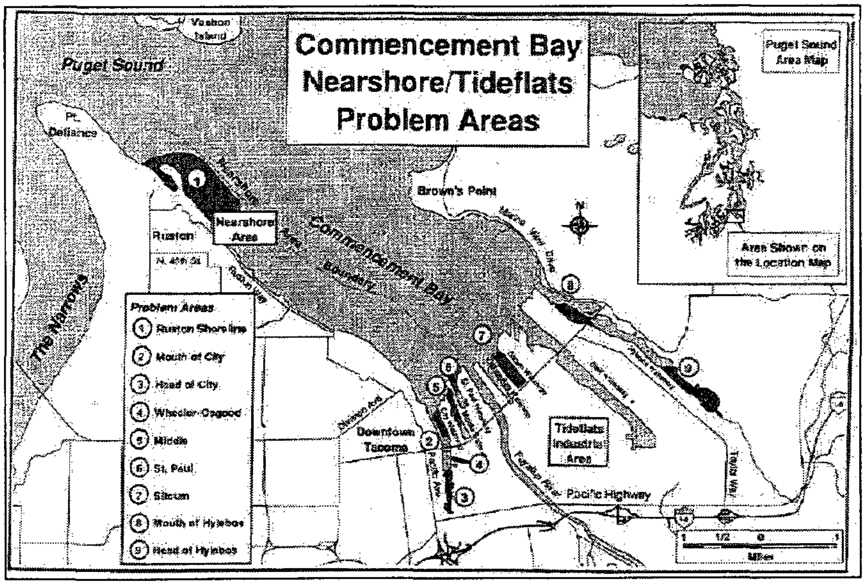


Figure 4.2. Delineation of Commencement Bay Nearshore area and location of problem areas (USEPA, 2002b).

The CBNT site is an industrialized commercial seaport covering approximately 31 km^2 of shallow water, shoreline and adjacent land. The land is occupied by various industrial, manufacturing and commercial operators, with lumber, pulp and paper, chemical and petroleum companies making up the area's major industries. An aerial photograph of the CBNT site is shown in Figure 4.3.

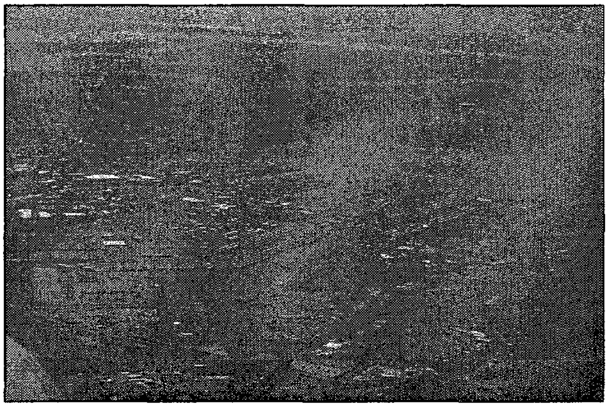


Figure 4.3. Aerial photograph of the CBNT site (USEPA, 2002b).

On-site contamination has been caused by both point and nonpoint sources. Early industrial surveys indicated that there are more than 281 industrial facilities in the area. Storm drains, contaminated groundwater, atmospheric deposition, direct dumping or filling, and industrial spills have all contributed to the site contamination as well. The local health department identified several hundred nonpoint sources that empty into Commencement Bay (USEPA, 2000).

Industries that may have historically contributed to contamination of MWW include metal foundries and machine works, shipbuilding, wood and coal tar products, metal scrap yards and industrial laboratories. The site currently hosts tugboat and marine transport operations, ship repair activities, warehousing and storage, and log rafting.

Industrial pollution in Commencement Bay area has resulted in deterioration of water and sediment quality. Contaminants found in problem areas include arsenic, cadmium, copper, lead, mercury, zinc, and various organic compounds such as polychlorinated biphenyls (PCB) and polycyclic aromatic hydrocarbons (PAH), and phthalates (USEPA, 2000). Within MWW the primary contaminants of concern are mercury, copper and PAH (USEPA, 2002b).

The case study will look at remediation efforts starting at the mouth of the MWW and extending approximately 700 m towards the head. This section of the waterway, shown in Figure 4.4, is under continual water cover.

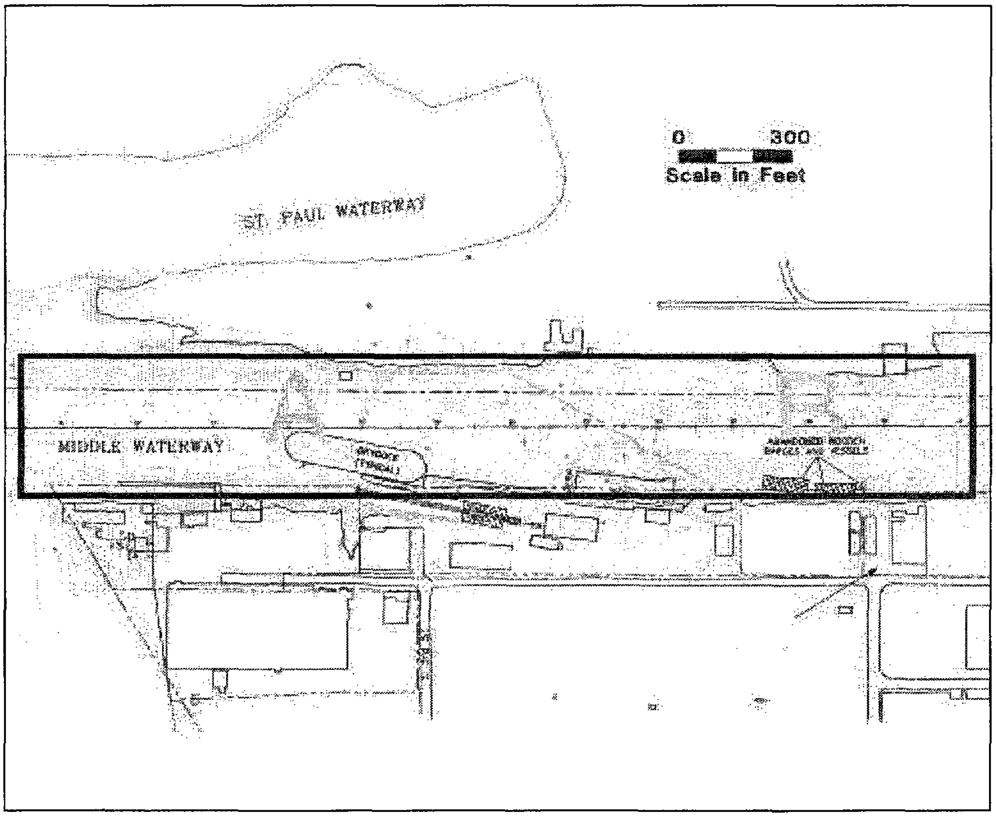


Figure 4.4. Section of Middle Waterway evaluated in the present study (Anchor & Foster Wheeler, 2001).

Sampling campaigns were carried out in May, 1988 and September, 1999 by Anchor Environmental for the Middle Waterway Action Committee in order to evaluate the extent of sediment contamination in MWW and provide a base for assessing possible remedial designs. Results indicated that the shallow and deep sediments were contaminated, up to depths of 3 m in some cases, by metals, PCB and PAH, in many cases in exceedance of sediment quality objectives (SQO) that had been set for the CBNT site by the USEPA (Anchor & Foster Wheeler, 2001). Chemical composition of the samples was varied and SQO exceedances did not demonstrate any spatial pattern. Particularly high concentrations of mercury, copper and PAH were observed. A list of contaminants and their SQO is given below in Table 4.1.

Table 4.1. Sediment Quality Objectives set for the Commencement Bay Nearshore/Tideflats Superfund project (Anchor & Foster Wheeler, 2001).

Compound	**SQO**	**Units**	**Compound**	**SQO**	**Units**
1,2-Dichlorobenzene	50	ug/kg	Di-n-octylphthalate	6200	ug/kg
2,4-Dimethylphenol	29	ug/kg	Fluoranthene	2500	ug/kg
Acenaphthene	500	ug/kg	Fluorene	540	ug/kg
Acenaphthylene	1300	ug/kg	Indeno(1,2,3-cd)pyrene	690	ug/kg
Anthracene	960	ug/kg	Naphthalene	2100	ug/kg
Benzo(a)anthracene	1600	ug/kg	N-Nitrosodiphenylamine	28	ug/kg
Benzo(a)pyrene	1600	ug/kg	PCB (Total)	300	ug/kg
Benzo(g,h,i)perylene	720	ug/kg	Pentachlorophenol	360	ug/kg
Benzofluoranthene	3600	ug/kg	Phenanthrene	1500	ug/kg
Benzoic acid	650	ug/kg	Phenol	420	ug/kg
Benzyl alcohol	73	ug/kg	Pyrene	3300	ug/kg
Bis(2-Ethylhexyl)phthalate	1300	ug/kg			
Butylbenzylphthalate	900	ug/kg	Antimony	150	mg/kg
Chrysene	2800	ug/kg	Arsenic	57	mg/kg
DDD	16	ug/kg	Cadmium	5.1	mg/kg
DDE	9	ug/kg	Copper	390	mg/kg
DDT	34	ug/kg	Lead	450	mg/kg
Dibenzo(a,h)anthracene	230	ug/kg	Mercury	0.59	mg/kg
Dibenzofuran	540	ug/kg	Nickel	140	mg/kg
Di-n-butylphthalate	1400	ug/kg	Zinc	410	mg/kg

4.1.2 Physical characterization of MWW

The following section describes physical characteristics of MWW measured during the 1998 and 1999 sampling campaigns. Unless otherwise noted, all data are taken from the Middle Waterway Final Data Evaluation Report (Anchor & Foster Wheeler, 2001).

Current in Middle Waterway was characterized via comparison to similar waterways in Commencement Bay (Hylebos and Thea Foss), in which long-channel currents dominate, however significant cross-channel currents and eddies exist. Net inflow occurs at the surface and in deep water with a balancing net outflow at mid-depths. Bottom current speeds near the mouth of Hylebos Waterway in a section similar to Middle Waterway can be in the range of 10 cm/s or greater.

Circulation and transport of particulate matter within the Commencement Bay waterways are not well established. A settling rate of 3 cm/year for Middle Waterway was calculated from average settling rates in neighbouring Hylebos, Sitcum and Thea Foss waterways measured using sediment traps. Burial rates were estimated using lead-210 core analyses, giving an average value of 1.2 cm/year. A resuspension rate of 1.8 cm/year was taken as the difference between the settling and burial rates. Deposition of particulate matter might be favoured due to the limited current, however boat traffic at the mouth of the waterway likely results in resuspension of sediments. Given reported settling velocities in the range of 2.5 m/day for organic and clay particles (O'Connor, 1988, cited in Chapra, 1997), it seems clear that significant resuspension is occurring in Middle Waterway. An average total suspended solids (TSS) value of 9.2 mg/L was measured in Middle Waterway.

Grain size data for shallow and deep sediments indicated that the sediments in Middle Waterway are primarily silty sands. Average organic carbon content for the water column and surface sediments was determined to be 3.8% and 2.5% for the subsurface sediments.

4.1.3 Remediation Options

Remediation options considered for Middle Waterway included various combinations of in-place capping, confined aquatic disposal, nearshore disposal, upland disposal, enhanced natural recovery, and monitored natural attenuation. This case study will evaluate two options for the relevant section of Middle Waterway: (i) monitored natural attenuation (MNA) and (ii) dredging and upland disposal of the contaminated sediments in a confined disposal facility. Descriptions of these management options are given in Section 2.1 of the Literature Review, with additional detail provided below.

4.1.4 LCA Goal and Scope

4.1.4.1 Purpose

The life cycle assessment conducted for this study seeks to quantitatively estimate and compare the environmental impacts associated with two options for managing contaminated sediments in the Middle Waterway of the Commencement Bay Nearshore Tideflats site. The purpose for conducting this study is to determine the least environmentally noxious option amongst the management plans presented. The LCA is intended for an academic audience to test the hypothesis indicated in Section 3.1.

4.1.4.2 Function

The function of the comparative study is the management of contaminated sediments so as to minimize the resulting impact to the environment.

4.1.4.3 Functional Unit

The functional unit for the LCA is the management for 50 years of approximately 71,000 m^3 of sediments contaminated with PAH, PCB and metals. The study area measures approximately 100 m by 700 m (USEPA, 2002a). The functional unit used here follows the recommendation of Diamond *et al.* (1999) for setting a LCA functional unit for a contaminated site management project, that being to use a physical measurement of the site requiring management.

4.1.4.4 Reference Flow

The reference flow will be the same for both options considered in this study: approximately 71,000 m^3 of contaminated sediments.

4.1.4.5 Remediation Scenarios

Two management scenarios were evaluated for the present study: (i) monitored natural attenuation (MNA), and (ii) dredging and disposal of the contaminated sediments within a confined disposal facility. The system of processes studied for each option are simplified representations, focussing on unit processes judged responsible for the majority of each option's impact. Technical details regarding the management options were approximated using generic descriptions in the literature and certain simplifying assumptions were made for each scenario due to the lack of data. Technical details and assumptions are provided below. It is assumed that the management options, as described here, respect technical feasibility and regulatory requirements, within the context of a remediation project located in Washington State, USA.

4.1.4.6 Option 1: Monitored natural attenuation

Option 1 consists primarily of sampling operations and transporting sediment samples for laboratory analysis over a 50 year period. Figure 4.5 shows a simplified system of processes for Option 1. Specific details regarding this management option are given in Appendix 2.

As mentioned above, the feasibility of this management option is assumed, however it should be noted that practically, the feasibility and appropriateness of this option is uncertain. A LCA does not constitute a comprehensive risk assessment, such as would be typically required to ensure that the resultant impacts are acceptable. Moreover, this option may not be acceptable to stakeholders, who may demand a higher standard of environmental quality, regardless of risk assessment results.

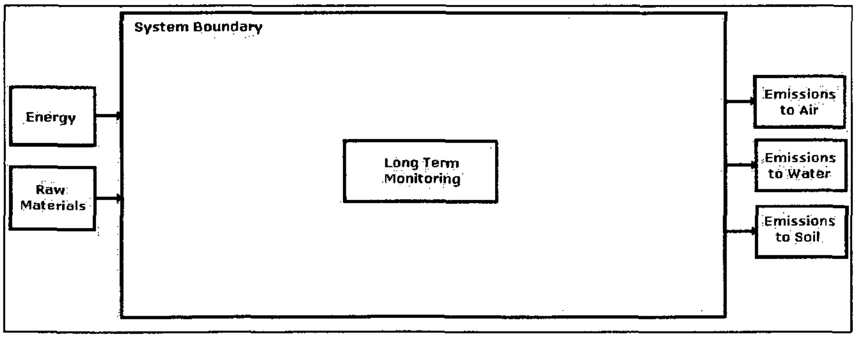


Figure 4.5. System of processes for Option 1, monitored natural attenuation.

4.1.4.7 Option 2: Dredging with disposal in a confined disposal facility

Option 2, dredging and disposal in a confined disposal facility, consists of site preparation, dike construction with clay liner, dredging, disposal, site closure and long-term monitoring. Figure 4.6 shows a simplified system of processes for Option 2. Specific details regarding this management option are given in Appendix 2. A schematic of a confined upland disposal facility is given in Figure 4.1 of the Literature review. For the sake of brevity, Option 2 will be referred to as "the Remediation Activities".

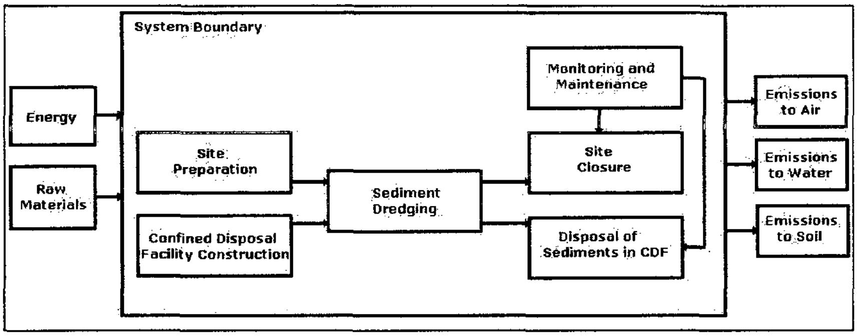


Figure 4.6. System of processes for Option 2, dredging and disposal within a confined disposal facility (CDF).

4.1.4.8 Time frame

The default time period over which remedial options will be evaluated is 50 years. Diamond *et al.* (1999) originally suggested 25 years as a temporal boundary, however other studies have found this to be insufficient, owing to extended length of time required for remediation (Cadotte *et al.*, 2007; Godin *et al.*, 2004; Bayer & Finkel; 2006). The time period of 50 years refers to the length of time for which the site will be managed, but this does not exclude evaluating emissions associated with the site beyond this time period for the monitored natural attenuation option.

4.1.4.9 Excluded processes

The chemicals required to analyze the monitoring samples collected for each option have not been evaluated as part of the present study, however it is recognized that the production and disposal of these chemicals could generate significant impacts.

Impacts due to MWW contaminants for which insufficient parameter data were available were not evaluated in the present study. A list of these parameters is listed in Table 4.2 below.

Table 4.2. Contaminants present at MWW excluded from the present study due to lack of sufficient parameter data.

Compound
2-Methylnaphthalene
4-Methylphenol
Acenaphthylene
Benzo(g,h,i)perylene
Benzoic acid
Benzyl alcohol
Bis(2-Ethylhexyl)phthalate
Dibenzofuran
Phenanthrene

The infrastructure surrounding the CDF, such as the roads and services to manage the facility have not been evaluated here. It is also assumed that the effluent produced by the CDF does not produce any impact within the receiving body of water (Great Lakes Commission, 2003; cited in Olsta, 2003).

4.1.5 Inventory

As stated above, technical details regarding the management options were approximated using generic descriptions in the literature. Certain data regarding construction vehicles (e.g., typical uses, cycle times) were taken from the LCI documentation of a previously performed remediation project LCA (Godin, 2004). These sources are noted in Appendix 2.

LCI data for construction activities were supplemented using the emission factors from the process emission database Nonroad (USEPA, 2008). The life cycle database Ecoinvent (Frischknecht, 2005) available in the software SimaPro 7.1.8 (Pre

Consultants, 2008) was used to expand the management option life cycle inventories. For example, the LCI data behind a kilogram of gravel or diesel consumed for construction purposes was determined using SimaPro.

For the MNA option, a mass balance model was used to evaluate contaminant concentrations in the water and sediment over time as well as to estimate off-site emissions via air and water. The time-integrated contaminant concentration in the sediments was entered into the LCI to calculate the primary impact to the sediments. Details regarding the calculation of this impact are given in Section 4.1.6.2. The mass of off-site emissions from the contaminated site were also calculated using the MNA mass balance model. This mass was entered into the LCI as an emission to air or water, as appropriate.

4.1.5.1 Initial Sediment Contaminant Concentrations

Sediments in MWW were divided into sediment units, shown if Figure 4.7, roughly delineated by the levels of contamination (Anchor & Foster Wheeler, 2001). Contaminant concentrations in the mixed (0 to 10 cm) and deep (10 to 30 cm) layers were averaged for each sediment unit. The averaged contaminant concentrations for the mixed and deep sediment layer are given in Table 4.3.

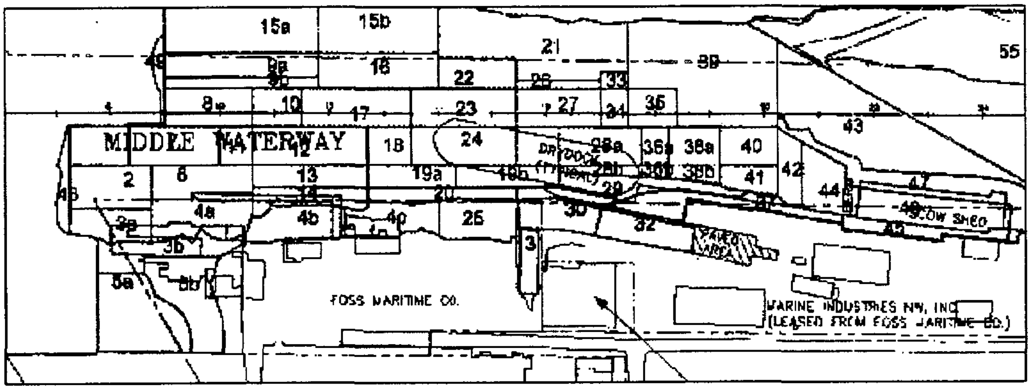


Figure 4.7. MWW sediment units used for calculating average sediment concentrations.

Table 4.3. Initial average contaminant concentrations in the mixed and deep sediments of MWW. Calculated from data in Anchor & Foster Wheeler (2001).

Substance	**Units**	**Mixed sediments concentration**	**Deep sediments concentration**
1,2-Dichlorobenzene	µg/kg	1.15E-01	0.00E+00
2,4-Dimethylphenol	µg/kg	3.11E+00	1.39E+01
Acenaphthene	µg/kg	1.64E+02	5.83E+02
Anthracene	µg/kg	4.07E+02	7.98E+02
Benzo(a)anthracene	µg/kg	9.28E+02	1.21E+03
Benzo(a)pyrene	µg/kg	7.62E+02	8.77E+02
Benzo(b)fluoranthene	µg/kg	1.36E+03	1.58E+03
Butyl benzyl phthalate	µg/kg	3.16E+00	0.00E+00
Chrysene	µg/kg	1.02E+03	1.35E+03
DDD	µg/kg	1.42E-01	2.91E+00
DDE	µg/kg	1.47E+00	1.57E+00
DDT	µg/kg	2.37E+00	5.97E+00
Dibenz(a,h)anthracene	µg/kg	3.15E+01	1.08E+02
Di-n-butyl phthalate	µg/kg	1.24E+01	0.00E+00
Di-n-octyl phthalate	µg/kg	1.18E+00	0.00E+00
Fluoranthene	µg/kg	2.14E+03	3.63E+03
Fluorene	µg/kg	2.29E+02	9.31E+02
Indeno(1,2,3-c,d)pyrene	µg/kg	2.39E+02	3.49E+02
Naphthalene	µg/kg	3.76E+02	6.43E+02
N-Nitrosodiphenylamine	µg/kg	2.13E+00	0.00E+00
PCB (total)	µg/kg	7.96E+01	8.92E+01
Pentachlorophenol	µg/kg	1.34E+01	1.67E+00
Phenol	µg/kg	7.51E+01	6.50E+01
Pyrene	µg/kg	1.82E+03	2.83E+03
Antimony	mg/kg	6.53E+00	2.71E+01
Arsenic	mg/kg	2.98E+01	2.64E+01
Cadmium	mg/kg	2.47E-01	4.24E-01
Copper	mg/kg	2.66E+02	3.37E+02
Lead	mg/kg	1.47E+02	1.38E+02
Mercury	mg/kg	1.37E+00	1.58E+00
Nickel	mg/kg	1.43E+01	1.14E+01
Zinc	mg/kg	2.12E+02	1.72E+02

4.1.5.2 Emissions calculated using MNA model

For the MNA management option, time profiles of chemical concentrations in Middle Waterway sediments and off-site emissions are calculated using a mass balance model of the contaminated sediments and the overlying water column. The theory and methods behind developing such models are described by Mills *et al.* (1982), Chapra (1997) and Ruiz *et al.* (2001).

The model consists of three compartments: a well-mixed water column, a surface layer of sediments, and a deeper layer of sediments. The surface layer is identified as mixed sediments, referring to the bioturbation and mechanical mixing action that this compartment experiences and the resulting interaction with the water column (Chapra, 1997; Ruiz *et al.*, 2000), however both the surface layer and deep sediments are modeled here as homogeneous volumes and therefore considered well-mixed.

The water column is given an average height of 2.58 m. Mixed sediments are modeled with a depth of 10 cm. This depth, describing where the bulk of benthic biological activity is located, is considered a 'worldwide' average, taken from a large database of marine and aquatic sediment mixing parameters (Boudreau, 1998). The deep sediments are modeled with a depth of 20 cm.

Figure 4.8 provides a graphic representation of the model. Within each of the three compartments, contaminants can exist in either a dissolved or particulate phase, according to their partitioning coefficients, which describe reversible sorption kinetics in a linear fashion. Mass transfer between the compartments is controlled by the processes of diffusion, settling, resuspension, and burial. Contaminants can be removed from the system via first order decay in each compartment or volatilization from the water column. Mass can enter or leave the system via an inflow and outflow.

The MNA model does not consider external contaminant loadings from other contaminated sites, the atmosphere or sewer discharges.

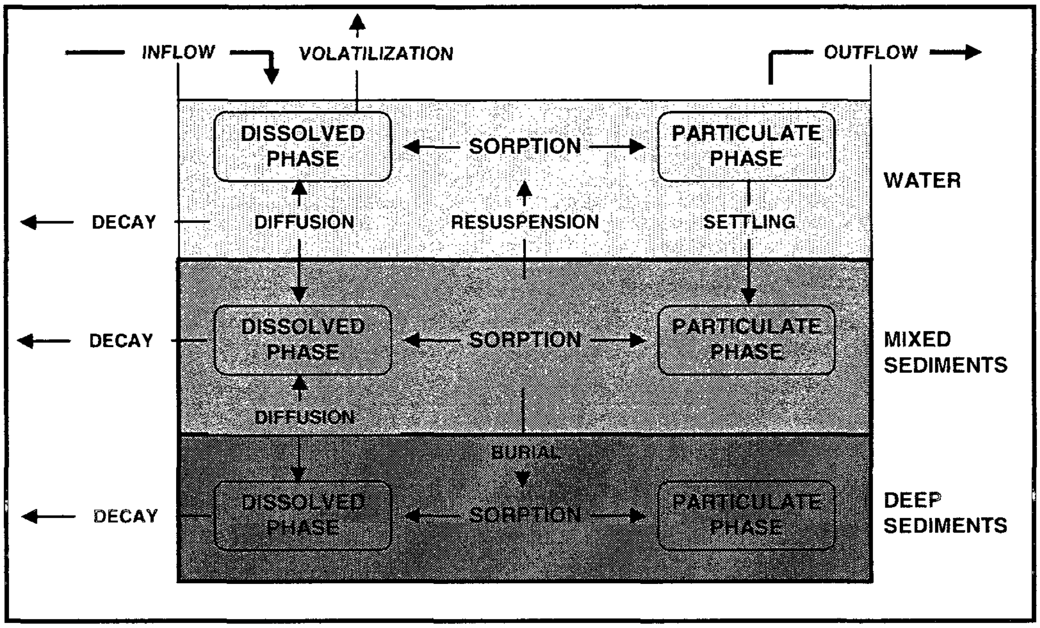


Figure 4.8. Graphic representation of MNA mass balance model.

The following section describes each mass balance, which calculates the change in mass over time for one contaminant in the relevant compartment. It is assumed that the mass transfer of each contaminant occurs independently of other contaminants.

4.1.5.2.1 Contaminant mass balances

The contaminant mass balance for the water column can be expressed in words as indicated in Equation 5.1 and symbolically as in Equation 5.2:

mass change over time = inflow – outflow – decay – volatilization – settling
+ resuspension + diffusion from sediments (Eq. 5.1)

$$\begin{aligned} V_w \cdot d(\frac{C_w}{dt}) &= Q \cdot C_{CB} - Q \cdot C_w - k_w \cdot V_w \cdot C_w - k_v \cdot V_w \cdot C_w \\ &- \nu_s \cdot A_w \cdot F_{pw} \cdot C_w + \nu_r \cdot A_m \cdot C_m \\ &+ \nu_d \cdot A_m \cdot (F_{dp} \cdot C_m - F_{dw} \cdot C_w) \end{aligned} \qquad \text{(Eq. 5.2)}$$

The contaminant mass balance for the mixed sediments layer can be expressed in words as indicated in Equation 5.3 and symbolically as in Equation 5.4:

mass change over time = settling – resuspension – decay – burial to deep sediments
+ diffusion from water + diffusion from deep sediments
(Eq.5.3)

$$\begin{aligned} V_m \cdot d(\frac{C_m}{dt}) &= \nu_s \cdot A_w \cdot F_{pw} \cdot C_w - \nu_r \cdot A_m \cdot C_m - k_s \cdot V_m \cdot C_m - \nu_b \cdot A_m \cdot C_m \\ &+ \nu_d \cdot A_m \cdot (F_{dw} \cdot C_w - F_{dp} \cdot C_m) + \nu_d \cdot A_m \cdot (F_{dp} \cdot C_{ds} - F_{dp} \cdot C_m) \end{aligned} \qquad \text{(Eq. 5.4)}$$

The contaminant mass balance for the deep sediments layer can be expressed in words as indicated in Equation 5.5 and symbolically as in Equation 5.6:

mass change over time = burial from mixed sediments – burial to deeper sediments
– decay + diffusion from mixed sediments (Eq. 5.5)

$$\begin{aligned} V_{ds} \cdot d(\frac{C_{ds}}{dt}) &= \nu_b \cdot A_m \cdot C_m - \nu_b \cdot A_m \cdot C_{ds} - k_s \cdot V_{ds} \cdot C_{ds} \\ &+ \nu_d \cdot A_m \cdot (F_{dp} \cdot C_m - F_{dp} \cdot C_{ds}) \end{aligned} \qquad \text{(Eq. 5.6)}$$

where:

V_w = volume of the water column [m^3]

V_m = volume of the mixed sediments layer [m^3]

V_{ds} = volume of the deep sediments layers [m^3]

A_w = surface area of water column [m^2]

A_m = surface area of mixed sediments [m^2]

C_w = contaminant concentration in Middle Waterway water [mg/m^3]

C_m = contaminant concentration in the mixed sediments [mg/m^3]

C_{ds} = contaminant concentration in deep sediments [mg/m^3]

C_{CB} = contaminant concentration in Commencement Bay water [mg/m^3]

Q = flow rate [m^3/day]

k_w = contaminant decay rate in water [1/day]

k_s = contaminant decay rate in sediments [1/day]

k_v = contaminant volatilization rate [1/day]

v_s = particulate matter settling rate [m/day]

v_r = sediment resuspension rate [m/day]

v_b = sediments burial rate [m/day]

v_d = diffusion mass-transfer coefficient [m/d]

F_{pw} = contaminant fraction in water in particulate form [-]

F_{dw} = contaminant fraction in water in dissolved form [-]

F_{dp} = contaminant fraction in sediments in dissolved form [-]

The following sections describe the processes modeled in the MNA model and the parameters that describe them. Data sources, assumptions and calculation details are given in the Appendices.

4.1.5.2.2 Physical parametrization of Middle Waterway

The majority of data for the physical characterization of MWW comes from the Middle Waterway Final Data Evaluation Report (Anchor & Foster Wheeler, 2001). A summary of physical characteristics of MWW used in the MNA model are given in Table 4.4. Unless otherwise specified with a superscript, values were taken from Anchor & Foster Wheeler (2001). Assumed and calculated parameter values are discussed below.

Table 4.4. Middle Waterway physical parameters.

Parameter	**Units**	**Value**
Water Column		
Average water volume, V_w	m^3	3.42E+05
Water volume at low tide	m^3	1.57E+05
Water volume at high tide	m^3	5.26E+05
Water volume difference between tides	m^3	3.69E+05
Return-flow factor, b[1]	-	0.431
Tidal period, T[2]	h	12.42
Effective flow rate, Q	m^3/day	3.08E+05
Average water depth	m	2.58
Surface area of water	m^2	9.18E+04
Average water temperature[3]	K	283.7
Organic carbon content in suspended sediment	%	3.80
Wind speed		
Average air temperature[4]	K	283.8
Average wind speed[5]	m/s	3.6
Particulate matter		
Settling rate	m/year	0.030
Burial rate	m/year	0.018
Resuspension rate	m/year	0.012
Sediments		
Surface area of mixed and deep sediments	m^2	9.18E+04
Depth of mixed and deep sediments[6]	m	0.10
Volume of mixed sediments	m^3	9.18E+03
Average sediment porosity	-	0.51
Average density of sediments	kg/m^3	2630
Organic carbon content in mixed sediments	%	3.80
Organic carbon content in deep sediments	%	2.50
Mixed sediments porosity	-	0.69
Deep sediments porosity	-	0.68

1 Calculated according to Sanford *et al.* (1992)

2 Lincoln (2000)

3 Calculated using data from National Oceanographic Data Center (2009)

4 Calculated using data from Western Regional Climate Center (2009)

5 Calculated using data from Windfinder (2009) for the Seattle-Tacoma Airport

6 Assumed

4.1.5.2.3 Waterway Surface Area and Volume

MWW surface area and volume were calculated using the bathymetric maps available in Anchor & Foster Wheeler (2001) which indicate the shoreline position of the mean lower low water (MLLW) and the mean higher high water (MHHW) as well as the bottom depth below MLLW across the waterway floor. Mean water volume in Middle Waterway was taken as the average water volume between high tide and low tide.

4.1.5.2.4 Water Flow and Advective Transport

Calculating the water flow in Middle Waterway requires tidal data and an estimate of the quantity of water that returns to the waterway with each tide.

Bathymetric maps (Anchor & Foster Wheeler, 2001) were used to calculate the difference in water volume between high tide and low tide. Puget Sound experiences a tidal period of 12 hours 25 minutes (Lincoln, 2000). As a rough first estimation, one could approximate the flow in Middle Waterway as the volume difference between the tides divided by the tidal period. However, a certain quantity of contaminant that is washed out into Commencement Bay returns to Middle Waterway with each tide.

The fraction of water that returns to a tidal embayment after flowing out into the receiving water body is described by the return flow factor, **b**, developed by Sanford *et al.* (1992) as part of a tidal flushing model. The model combines theory from the tidal prism concept for estimating water quality in an estuary or embayment (Schwartz, 2005) with that of horizontal jet flow (Carter, 1969; cited in Sanford *et al.*, 1992). The full details of the model will not be discussed here. The relevant calculations for determining the return flow factor are discussed in Appendix 3.

Using data from nearby waterways where Middle Waterway data were not available, a return flow factor of **b** = 0.431 was calculated (see Appendix 3), indicating that 57% of the tidal volume can be considered as outflow. Using the return flow factor, the volume difference between tides and the tidal period, an effective flow rate of 3.08E+05 m^3/day for Middle Waterway was calculated (see Appendix 3).

It should be noted that flow from Middle Waterway to Commencement Bay is modeled entirely according to the astronomical tides. Mixing as a result of wind action or flushing resulting from precipitation has not been modeled here. Middle Waterway receives inflow from several storm drains which have also not been included in the model. It is assumed that tidal flow is the dominant advective force in Middle Waterway (Sanford *et al.*, 1992).

For the sake of this model, the contaminant concentration in Commencement Bay is assumed to be zero and, as indicated above, no external contaminant loading is accounted for. Given that the Commencement Bay Superfund Site consists of nine problem areas located in an industrial harbour, this assumption is likely to be incorrect. However, water quality data for Commencement Bay were unavailable and it is beyond the scope of this project to model the water quality of Commencement Bay due to the presence of other contaminated sites.

The Commencement Bay Superfund remediation project was initiated to improve the environmental quality of the sediment and water in Commencement Bay. Actions initiated by the project include direct interventions, such as those considered for Middle Waterway, as well as "institutional controls" that attempt to halt contaminant loading from surrounding industries via regulation or information campaigns (USEPA, 1989). It can be assumed that these interventions will be successful to a certain degree and that contaminant concentrations in Commencement Bay will decrease over time.

A modeling limitation also necessitates the assumption of zero contaminant loading from Commencement Bay, in that if the inflow concentration to Middle Waterway is set as a non-zero constant, this concentration tends to drive the model results in both the water column and sediments.

Perhaps the most important justification for discounting external contaminant loading is that the system boundaries of the LCA are limited to the remediation of the MWW site. Impacts determined using the MNA model are meant to reflect decisions taken regarding the MWW site only and the inclusion of external contaminant loadings, while more physically correct, would bias the LCA results.

4.1.5.2.5 Settling, Burial and Resuspension

Particulate matter is removed from the water column by settling under the force of gravity. Bioturbation caused by benthic organisms and mechanical mixing by water will return a certain portion of the sediments to the water column. Over time, particulate matter will accumulate in the sediment layer and be buried.

Settling rates are applied within the water mass balance to the contaminant fraction in the particulate phase, however resuspension and burial are applied to both the particulate and dissolved fractions, accounting for transport of both phases (Chapra, 1997; Ruiz, 2001).

Under the forces of compaction associated with sediment burial, porewater can be transported both horizontally and vertically into adjacent compartments (Mulligan & Yong, 2006). This transport mechanism is not accounted for in the MNA model and the extent to which this exclusion underestimates contaminant loading from deeper sediments is unknown.

The settling rate for MWW was calculated from average settling rates in neighbouring Hylebos, Sitcum and Thea Foss waterways which were measured using sediment traps. Burial rates were estimated using lead-210 core analyses (Anchor & Foster Wheeler, 2001). The resuspension rate was taken as the difference between the settling and burial rates (Anchor and Foster Wheeler, 2001). Values are presented in Table 4.4 above.

4.1.5.2.6 Substance-specific data used in the MNA model

Substance-specific data used in the MNA model are listed in Table 4.5. These data come primarily from the CalTOX (McKone *et al.*, 2001) and USEtox (Rosenbaum *et al.*, 2008) databases and are used to calculate decay rates, volatilization rates, partitioning coefficients and diffusion mass transfer coefficients. The parameter values presented here were measured under conditions that may not represent those of the MWW site, however it is assumed that the error in using these values is not significant.

Table 4.5 Substance-specific data used in the MNA model (McKone *et al.*, 2001; Rosenbaum *et al.*, 2008)

	Molecular Weight	Henry's law constant[1]	Organic carbon partition coefficient	Half-life in surface water	Half-life in sediments
	MW	H_e	K_{oc}	$T_{1/2}$ (w)	$T_{1/2}$ (s)
	g/mol	Pa-m^3/mol	-	d	d
1,2-Dichlorobenzene	146.97	2.04E+02	3.84E+02	1.04E+02	3.79E+02
2,4-Dimethylphenol	122	2.03E-01	1.80E+01	1.75E+00	2.29E+01
Acenaphthene	154.21	4.03E+01	5.03E+03	6.31E+00	6.31E+00
Anthracene	178.22	4.28E+00	2.22E+04	4.75E-02	9.45E+02
Benzo(a)anthracene	228.29	8.13E-01	4.59E+05	8.33E-02	1.41E+03
Benzo(a)pyrene	252.3	9.20E-02	2.49E+06	2.34E+00	1.17E+03
Benzo(b)fluoranthene	252.3	1.52E+00	9.17E+05	1.52E+01	1.40E+03
Butyl benzyl phthalate	312	1.69E-01	4.27E+04	4.00E+00	9.05E+01
Chrysene	228.3	1.59E-01	2.71E+05	3.60E-01	2.19E+03
DDD	320.05	4.49E-01	6.48E+05	3.21E+03	2.88E+03
DDE	318.05	4.92E+00	8.64E+04	3.37E+00	2.86E+03
DDT	354.5	1.10E+01	3.09E+05	1.79E+02	2.86E+03
Dibenzo(a,h)anthracene	278.35	3.52E-03	2.03E+06	1.64E+01	2.01E+03
Di-n-butyl phthalate	278.35	1.63E-01	1.58E+03	7.50E+00	1.20E+01
Di-n-octyl phthalate	390.57	1.17E-01	1.57E+09	1.75E+01	1.86E+02
Fluoranthene	202.26	1.02E+00	4.94E+04	1.74E+00	9.50E+02
Fluorene	166.2	9.47E+00	9.75E+03	4.60E+01	1.36E+02
Indeno(1,2,3-c,d)pyrene	276.3	7.04E-03	8.43E+06	1.88E+02	1.76E+03
Naphthalene	128.16	4.67E+01	1.07E+03	1.03E+01	1.29E+02
N-Nitrosodiphenylamine	192	5.07E-01	1.20E+03	1.55E+01	2.29E+02
PCB (total)	340.82	1.81E+01	1.26E+06	2.77E+02	1.34E+04
Pentachlorophenol	266.5	2.53E-03	9.00E+02	8.26E-02	2.29E+02
Phenol	94	4.02E-02	1.60E+01	4.04E-01	2.29E+01
Pyrene	202.3	1.40E+00	7.06E+04	5.67E-02	3.91E+03

[1] Henry’s law constant reported for temperature range of 20°C to 25°C

* Decay rate [$second^{-1}$]. Data taken from USEtox (Rosenbaum *et al.*, 2008)

4.1.5.2.7 Decay rate constants

Half-lives used to calculate decay rate constants are listed in Table A2.4. Half-lives were taken from data available in the CalTOX (McKone *et al.*, 2001) and USEtox (Rosenbaum *et al.*, 2008) databases, with most of these data originally taken from

Howard *et al.* (1991), except for that of benzo(a)pyrene which was taken from Risk Science Program (1995). These data are generated by a variety of sources, representing a range of transformation removal processes. Typically, rates associated with photolysis, hydrolysis, oxidation/reduction, and microbial degradation are reported and an average value is used. It is unknown which processes dominate in MWW, and as such the averages of half lives or decay rates reported in CalTOX or USEtox were used for the model. The products resulting from transformation are not considered in the MNA model.

The decay rates used here have a high level of uncertainty associated with them. Besides experiencing natural variation that can span several orders of magnitude, the average rates are often based on a limited number of measurements, taken under conditions which are unlikely to be representative of those found in Middle Waterway. In many cases, scientific judgment has been used as a proxy for measurement (Howard *et al.*, 1991). Caution must be used when interpreting results based on these values.

Water and sediment decay rates, $\mathbf{k_w}$ and $\mathbf{k_s}$ [/day], are calculated from half-lives, $T_{1/2}$ [day], according to Equation 5.7:

$$k = \frac{\ln(2)}{T_{1/2}} \qquad \text{(Eq. 5.7)}$$

4.1.5.2.8 Volatilization

Volatilization acts on the dissolved portion of contaminants in the water column, according to the following term in Equation 5.2:

$$- k_v \cdot V_w \cdot C_w$$

The transfer from the dissolved to gaseous phase is an important removal process, particularly for low molecular weight PAHs (Chapra, 1997; Mills *et al.*, 1982). As a rule of thumb, volatilization is likely to be an important fate process for substances whose Henry's law constant is greater than 100 Pa-m^3/mol (Mills *et al.*, 1982).

Volatilization rates were calculated for the contaminants found in MWW sediments. These rates were used in the water column mass balance and to determine the quantity of pollutants emitted to air from the site. Calculated volatilization rates are listed in Appendix 4 , along with details of the calculations, performed according to the methods indicated in Mills *et al.* (1992), Chapra (1997) and Ruiz *et al.* (2001).

Henry's law constants used for the calculations were taken from the CalTOX and USEtox databases and are listed in Table A4.1. USEPA (1994) is the original source for most of these data, except for that of benzo(a)pyrene which was taken from Risk Science Program (1995).

An average wind speed of 3.6 m/s was calculated using an average of historical data for the Seattle-Tacoma Airport (Windfinder, 2009), located approximately 3 km to the east of Puget Sound and 22 km to the northeast of Middle Waterway. It is uncertain how representative these data are for Middle Waterway and their use introduces additional uncertainty to the results produced by the model.

4.1.5.2.9 Diffusion

The process of diffusion in the MNA model is driven by the dissolved phase concentration difference between compartments, acting across the surface area that separates them. Diffusion is represented in the mass balances by the following term:

$$\text{diffusion term} = v_d \cdot A \cdot (F_{da} \cdot C_a - F_{dl} \cdot C_l)$$

where:

v_d = diffusion mass-transfer coefficient [m/d]

F_{da} = dissolved contaminant fraction in adjacent compartment [-]

F_{dl} = dissolved contaminant fraction in compartment of interest [-]

A = surface area separating the two compartments [m^2]

The diffusion mass transfer coefficient at the sediment-water interface, or between sediment layers, $\mathbf{v_d}$ [m/yr – note different units], can be estimated with the porosity, $\boldsymbol{\varphi}$, and the substance's molecular weight, **MW** [g/mol], from the empirical relationship derived by Thomann and Mueller (1987, cited in Chapra, 1997)

$$v_d = 69.35 \cdot \varphi \cdot MW^{-2/3}$$

4.1.5.2.10 Contaminant Partitioning

Certain fate processes act on the dissolved phase of a substance, such as volatilization and diffusion, while other processes such as advection, decay, resuspension and burial affect both phases. Only the particulate phase experiences settling from the water column.

The dissolved and particulate fractions for each contaminant are primarily a function of the substance's partitioning coefficient and fraction of organic carbon available in the compartment of interest. Contaminant fractions in the particulate and dissolved phase were calculated for the water column and sediment layer according to the formulas outlined in Chapra (1997). Calculation details and data sources are discussed in Appendix 5.

4.1.5.3 Calculating LCI quantities for MNA primary impact assessment

The output of the MNA model are expressions for contaminant concentrations as a function of time for the mixed sediments, deep sediments and water column. Integration of these concentration expressions over time permitted the evaluation of the contaminant mass emitted from the site via water and air [kg], as well as the time-integrated sediment concentration [kg-year/m^3]. The mass emitted via water from the MWW site to Commencement Bay and Puget Sound was calculated by multiplying the integral of the water concentration expression by the flow rate over the time period evaluated. Similarly, multiplication of this integral by the average water volume and volatilization rate gave the mass emitted to air. The time-integrated contaminant concentration in the mixed sediments was calculated by multiplying the integral of the sediment concentration expression by the mixed sediments volume. These quantities were entered into the LCI of the MNA management option, evaluated over a period of 100 years. This time period was chosen to ensure that significant impacts would not be neglected and is considered to be at the limit of what can be predicted regarding site-usage. Other studies have used this time period to evaluate MNA (Newell *et al.*, 2002; Hausman & Rifai, 2005; Go *et al.*, 2009)

4.1.5.4 Sensitivity analysis

The sensitivity of MNA model results to various model and chemical parameters was tested by reducing parameter values by 10% and measuring the resulting change in model results. Sensitivity analyses were carried out on the following parameters: wind speed, sediment and suspended sediment organic carbon content, total suspended sediment, porosity, organic-carbon partitioning coefficient, Henry's law constant, water decay rate, sediment decay rate and the return flow factor.

4.1.5.5 LCI Ecosphere/Technosphere distinction

The LCI of Option 1 was evaluated using both the ecosphere and technosphere perspectives. Impacts due to on-site sediment contamination were only included in the ecosphere perspective. Off-site emissions from the MWW site were included in both the ecosphere and technosphere perspectives.

4.1.6 Impact Analysis

4.1.6.1 LCIA method

IMPACT 2002+ was used as the LCIA method for the present study. A description of this method is offered in Section 2.2.4 of the Literature Review. The model used by IMPACT 2002+ to determine the intake fractions and fate factors for toxicity and ecotoxicity characterization factors is available in spatial and non-spatial versions, thus making it well-suited to studying spatial differentiation. Modifications that have been made to the LCIA method for the purposes of this study are discussed in the following sections.

4.1.6.2 Sediment Ecotoxicity Effect Factors

A significant impact associated with the project is that caused by the contamination within the sediments. Within most LCIA methods, sediments are treated as a ‘sink’ and all substances that partition to sediments are not counted as having an impact. In order to evaluate this impact, ecotoxicity effect factors for sediments were calculated in a similar manner to soil ecotoxicity effect factors (Jolliet *et al.*, 2003) in IMPACT 2002+ and sediment ecotoxicity factors of the LCIA method USES-LCA (Huijbregts, 1999). Due to a lack of appropriate toxicity data for soil and sediments, effect factors for these media are calculated from aquatic effect factors using the Equilibrium Partitioning (EqP) method described in Di Toro *et al.* (1991). Equation 5.8 below describes the calculation for sediment ecotoxicity factors:

$$EF_{sed} = EF_{aq} / (K_{sed\text{-}aq} * \rho_{sed}) \qquad \text{(Eq. 5.8)}$$

where:

EF_{sed} = sediment effect factor [PAF per kg substance i/m^3 sediment]
EF_{aq} = aquatic ecotoxicity effect factor [PAF per kg substance i/m^3 water]
$K_{sed\text{-}aq}$ = sediment-water partitioning coefficient [m^3/m^3]
ρ_{sed} = sediment density [kg/L]

The EqP method originated as a means of calculating sediment quality criteria using water quality criteria (WQC) and equilibrium partitioning coefficients. The assumptions employed for the EqP method are that (i) the pore water and sediment phases are in equilibrium with each other, (ii) the dominant exposure route is via pore water and (iii) aquatic and sediment species are equally sensitive to environmental insults (Huijbregts, 1999). The ecotoxicity effect factors calculated here were applied to the top 10 cm of sediments, where the majority of benthic biological activity is located (Boudreau, 1998).

As impacts to sediment typically can not be inferred from chemistry data alone (Chapman *et al.*, 1987; Burton *et al.*, 2002; Chapman & McDonald, 2005; Chapman, 2007), the effect factors developed here were not applied to sediments outside of the MWW site, following status quo for treatment of sediments in most LCIA methods (Bare *et al.*, 2003; Jolliet *et al.*, 2003; Toffoletto *et al.*, 2007). However, environmental risk assessment results indicated that a significant impact was present at MWW site (USEPA, 1989; Anchor & Foster Wheeler, 2001) and as such, sediment ecotoxicity effect factors were applied. It is assumed that this inconsistency in the application of sediment ecotoxicity effect factors does not significantly underestimate the impact caused by the MNA option.

The calculation of on-site impacts is carried out by multiplying the sediment ecotoxicity effect factors by the time-integrated sediment concentrations calculated using the MNA model. The time-integrated concentrations take into account the time over which sediments have been impacted and have units of kg-year/m^3.

Multiplication of these values by the sediment ecotoxicity effect factors with units of PAF per kg*i*/m^3 and the volume of contaminated sediments [m^3] gives ecotoxicity impact values with units of PAF-m^3-year.

4.1.6.3 Toxicity & Ecotoxicty Characterization Factors

Human toxicity and ecotoxicity characterization factors were calculated with the IMPACT multimedia fate and exposure model using three levels of spatial differentiation for emissions to water and two levels of spatial differentiation for emissions to air, in all cases using the most appropriate available modeling zone to represent emissions from the MWW site. Impacts were evaluated with each set of characterization factors. LCIA results for the management options are presented according to three levels of spatial differentiation available with the IMPACT models: Non-spatial model, Watershed Model and Modified Watershed model.

4.1.6.3.1 Non-spatial model

The Non-spatial IMPACT model for Canada was used to evaluate water and air emissions occurring in Washington State. This model version is described in Section 2.2.4.2 of the Literature Review and model parameters are listed in Table A1.2 of Appendix 1.

4.1.6.3.2 Watershed model

For the Watershed model version, IMPACT North America was used, as description of which is provided in Section 2.2.4.2 of the Literature Review. Water emissions were modeled as occurring in the zone labelled WUS317, which represents HUC 171100 (Washington State) in the USGS Hydrologic Unit Code system. A map showing the location of this watershed is given in Figure A1.2 of Appendix 1 and parameter values for this zone are given in Table A1.2 of Appendix 1. Note that fish catch data for this zone were modified, as described below.

Air emissions were evaluated using the Urban box within IMPACT North America to represent Tacoma, Washington. Urban box parameters for Tacoma are available in Table A1.3 of Appendix 1.

4.1.6.3.3 Modified Watershed model

A second version of the IMPACT North America model was used for this project with an added watershed zone to represent Puget Sound, the body of water on which MWW is located. Water emissions were evaluated using this zone as the emission location. Parameter values for this zone are given in Table A1.2 of Appendix 1. With the creation of this zone, it was necessary to ensure that mass balance was conserved within the model, particularly with respect to advective flow between zones. Flow between Puget Sound and the ocean was subtracted from flow between zone WUS317 and the ocean. Additionally, the surface area of WUS317 intersecting the air zones W21, W22, X21 and X22 (see Figure A1.1 in Appendix 1) was recalculated so as not to include the area of Puget Sound.

Average fish harvest data from 1998-2007 for Puget Sound were obtained from the Washington Department of Fish and Wildlife (2009). The IMPACT North America model estimates the yearly edible fish catch for each watershed with extrapolated data from the Food and Agricultural Organization of the United Nations (FAO, 2005, cited in Humbert *et al.*, 2009). The two data sources did not agree and the WA Department of Fish and Wildlife data were used for both the Watershed model and the Modified Watershed model. In the latter model, the Puget Sound fish catch was subtracted from the fish catch attributed to WUS317 to avoid double-counting.

Air emissions were evaluated using the Urban box within IMPACT North America to represent Tacoma, Washington.

4.2 Calculation tools

Maple Version 10 (Maplesoft, 2005) was used to solve the systems of mass balance equations that made up the MNA model. SimaPro 7.1.8 Faculty Version (Pré Consultants, 2008) was used to carry out LCA calculations. Microsoft Excel 2002 (Microsoft Corporation, 2002) was used to run the IMPACT models and perform all additional calculations.

5 Results

This section presents the main results produced by the present study. Additional results are presented in Section 6, Complementary Results. It should be noted that the results are presented in a different order than the Methodology. Characterization factors produced by the modifications to the LCIA method are presented first, followed by the LCA results in which these characterization factors are applied.

5.1 Modifications to LCIA method

5.1.1 Sediment ecotoxicity Effect Factors

Presented in Table 5.1 are the aquatic ecotoxicity effect factors [PAF per kg/m^3] for contaminants found at MWW site, the corresponding sediment ecotoxicity effect factors [PAF per kg/m^3] and the sediment partitioning coefficients [m^3/m^3] used to calculate them. Also included for the purposes of comparison are the inverse values of the sediment quality objectives (SQO) set for the CBNT Superfund site.

With the exception of phenol and 2,4-dimethylphenol, all the substances considered have sediment partitioning coefficients greater than one, indicating a preference to stay attached to the sediment than to partition to water. As a result, most sediment ecotoxicity effect factors were less than their corresponding aquatic effect factor.

By means of comparison, Figure 5.1 presents the sediment ecotoxicity effect factors calculated in the present study plotted against the inverse of their respective SQO values. One could hypothesize that substances with a greater ecotoxicity effect factor would have a greater 1/SQO value, however no immediate correlation is evident. Of particular interest is the significant reduction in effect factor for highly hydrophobic contaminants with high aquatic effect factors and low SQO, such as benzo(b)fluoranthene, indeno(1,2,3-c,d)pyrene and PCB.

Table 5.1. Ecotoxicity effect factors for water and sediment, sediment partitioning coefficients (**K_d**) and the inverse of sediment quality objectives (**1/SQO**).

Substance	**Ecotoxicity EF**		**K_d**	**1/SQO**
	Aquatic	**Sediment**		
1,2-Dichlorobenzene	1.72E+00	1.66E-01	2.00E-02	1.04E+01
2,4-Dimethylphenol	7.28E+01	1.50E+02	3.45E-02	4.86E-01
Acenaphthene	4.15E+03	3.06E+01	2.00E-03	1.36E+02
Anthracene	3.92E+04	6.55E+01	1.04E-03	5.99E+02
Benzo(a)anthracene	2.45E+05	1.98E+01	6.25E-04	1.24E+04
Benzo(a)pyrene	2.27E+04	3.38E-01	6.25E-04	6.72E+04
Benzo(b)fluoranthene	2.31E+05	9.33E+00	2.78E-04	2.47E+04
Butyl benzyl phthalate	2.27E+03	1.98E+00	1.11E-03	1.15E+03
Chrysene	2.74E+05	3.75E+01	3.57E-04	7.31E+03
DDD	1.38E+05	7.90E+00	6.25E-02	1.75E+04
DDE	2.88E+04	1.23E+01	1.11E-01	2.33E+03
DDT	1.48E+05	1.78E+01	2.94E-02	8.33E+03
Dibenz(a,h)anthracene	1.25E+07	2.28E+02	4.35E-03	5.48E+04
Di-n-butyl phthalate	2.85E+03	6.69E+01	7.14E-04	4.26E+01
Di-n-octyl phthalate	1.50E+01	3.53E-07	1.61E-04	4.24E+07
Fluoranthene	1.22E+03	9.15E-01	4.00E-04	1.33E+03
Fluorene	2.60E+03	9.87E+00	1.85E-03	2.63E+02
Indeno(1,2,3-c,d)pyrene	1.70E+06	7.45E+00	1.45E-03	2.28E+05
Naphthalene	2.05E+03	7.13E+01	4.76E-04	2.87E+01
PCB (total)	5.25E+05	1.54E+01	3.33E-03	3.40E+04
Pentachlorophenol	9.01E+03	3.71E+02	2.78E-03	2.43E+01
Phenol	8.26E+01	1.91E+02	2.38E-03	4.32E-01
Pyrene	2.56E+04	1.34E+01	3.03E-04	1.90E+03
Antimony	9.00E+02	7.41E+01	6.67E-06	1.21E+01
Arsenic	1.81E+02	1.03E-01	1.75E-05	1.75E+03
Cadmium	2.62E+02	3.88E-04	1.96E-04	6.74E+05
Copper	1.47E+04	7.42E+01	2.56E-06	1.98E+02
Lead	1.16E+03	1.73E-03	2.22E-06	6.74E+05
Mercury	2.31E+04	4.20E+00	1.69E-03	5.51E+03
Nickel	5.78E+02	1.59E+01	7.14E-06	3.64E+01
Zinc	1.52E+03	3.65E+01	2.44E-06	4.18E+01

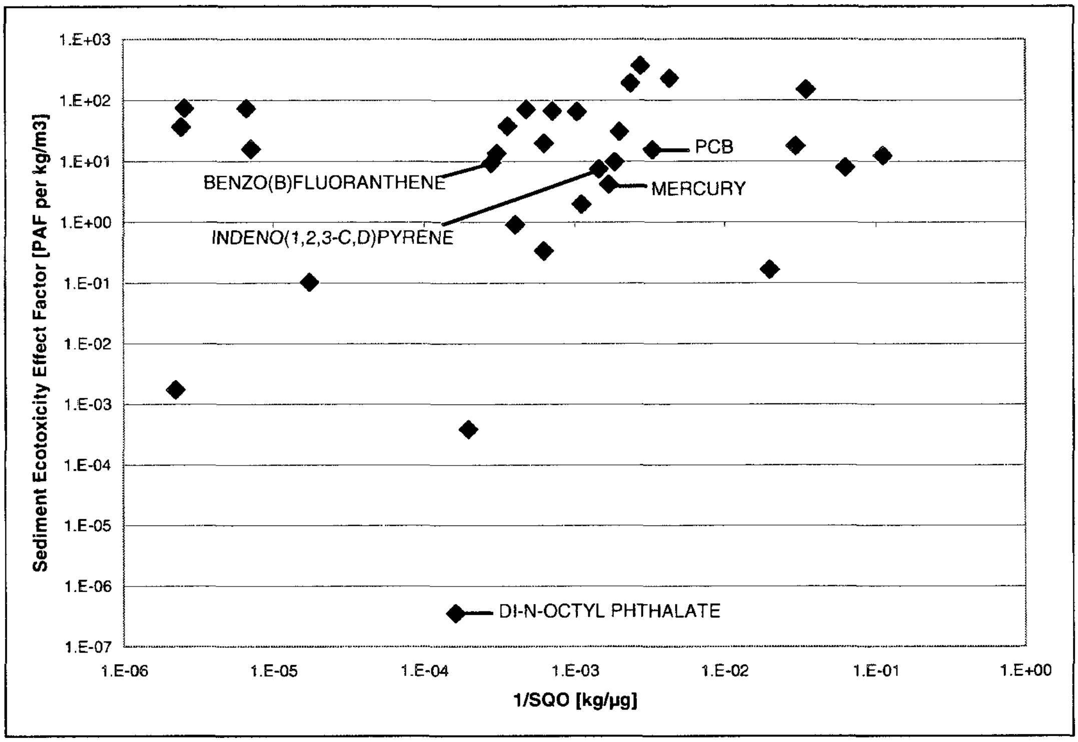


Figure 5.1. Calculated sediment ecotoxicity effect factors plotted against the inverse of the sediment quality objectives set for the CBNT Superfund site.

5.1.2 Toxicity and Ecotoxicty Characterization Factors

This section presents the human toxicity and ecotoxicity characterization factors produced by the various versions of the IMPACT model used in the present study. Results are separated according to the media in which the emission occurred.

5.1.2.1 Emission to water

Figures 5.2 and 5.3 present the ecotoxicity and human toxicity characterization factors produced by the three IMPACT model versions for an emission to water.

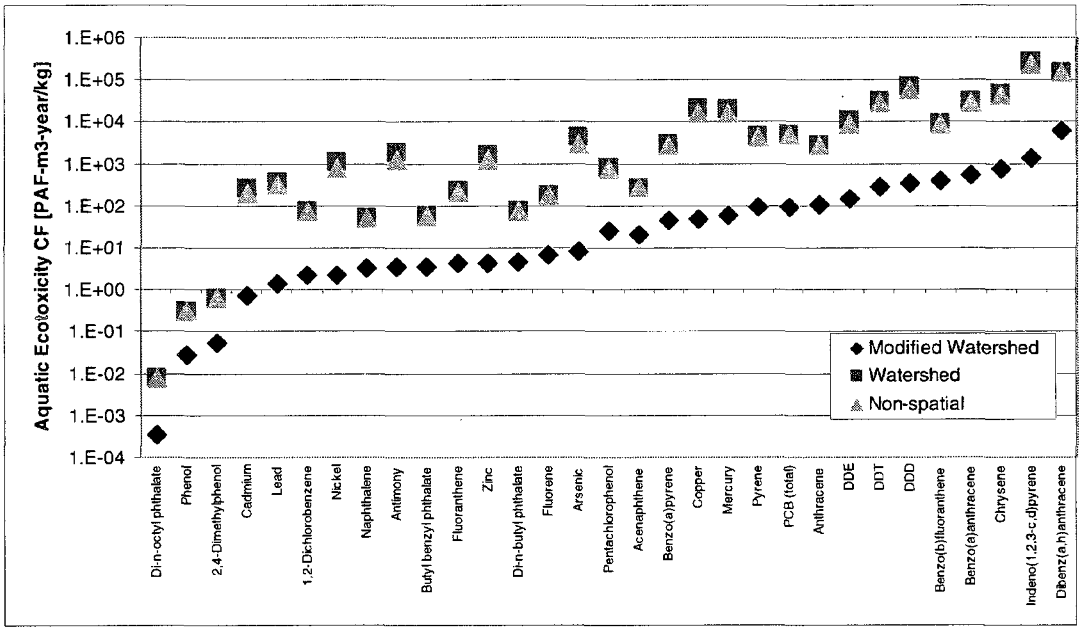


Figure 5.2. Aquatic Ecotoxicity characterization factors calculated using the Non-spatial, Watershed and Modified Watershed models.

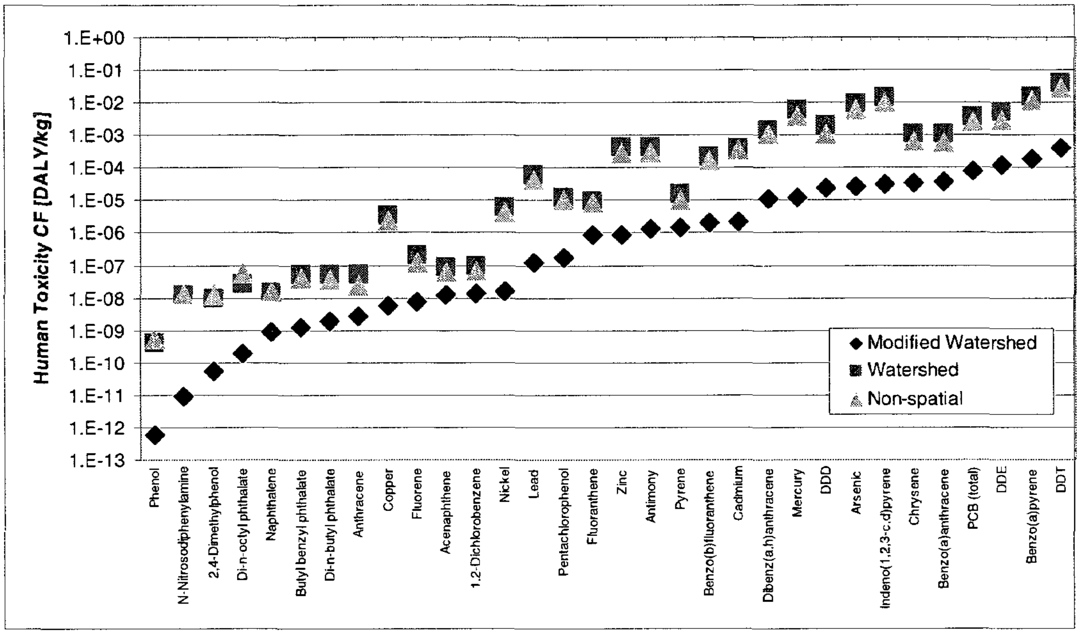


Figure 5.3. Human toxicity characterization factors calculated using the Non-spatial, Watershed and Modified Watershed models.

5.1.2.1.1 Ecotoxicity

Little difference exists between the ecotoxicity CF produced by the Non-spatial and Watershed models, but a difference of one or two orders of magnitude can be seen between ecotoxicity CF produced by the Watershed and Modified Watershed models. The greatest difference between the models was noted amongst the metals, which typically had two orders of magnitude difference between CF. Persistent organic substances (long half-lives in water, see Table A1.4 in Appendix 1) such as DDT, DDD and indeno(1,2,3-cd)pyrene also had two orders of magnitude difference between the ecotoxicity CF produced by the Watershed and Modified Watershed models. The smallest difference in CF between the model was for phenol (factor 10.9) and 2,4-dimethylphenol (factor 12.2).

Dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene and chrysene had the greatest ecotoxicity CF for an emission to water for the Modified Watershed model (5.89E+03, 1.39E+03 and 7.47E+02 PAF-m3-year/kg, respectively), having a combination of long half-life in water and high toxicity, expressed as a high aquatic ecotoxicological effect factor (see Table A1.4 of Appendix 1). The lowest ecotoxicty CF were for the substances di-n-octyl phthalate, phenol and 2,4-dimethylphenol, each with a low half-life in water and low aquatic ecotoxicological effect factor.

The different model versions also displayed different ecotoxicity rankings of the contaminants, with metals showing a significant decrease in ecotoxicity with the Modified Watershed version than with the Non-spatial and Watershed versions and dibenzo(a,h)anthracene and indeno(1,2,3-cd)pyrene switching first and second ranking.

The similarities and differences between the Non-Spatial, Watershed and Modified Watershed model results for ecotoxicity CF due to a water emission can be largely attributed to the water residence time in the zone where the emission occurs. The residence times of watershed HUC 171100 (1.84E+05 hours) and the Non-Spatial model (5.30E+05 hours) are similar compared to that of Puget Sound, which is two orders of magnitude lower (3.48E+03 hours). The fate factor (and thereby the CF) for a substance will vary directly with its residence time in the compartment of interest (Jolliet *et al.*, 2003). Persistent substances demonstrate this difference to a greater degree, as the full residence time is "utilized" before the substance is transferred to the ocean, where no further ecotoxicity impact is evaluated. Substances with a low half-life in water such as phenol and 2,4-dimethylphenol show less difference between the models, as the majority of the substance is degraded before being transferred out of the watershed zone and therefore exhibits the same impact wherever it is emitted.

Relative persistence can explain the modification of ecotoxicity ranking between the models, in that for the versions with higher water residence times, substances with long half-lives in water are given a longer amount of time to complete their exposure route. With the Modified Watershed version, substances emitted into Puget Sound have less time in water before they reach the ocean and as such, the relative toxicity of substances becomes more important. For example, while indeno(1,2,3-cd)pyrene has a higher ecotoxicity CF than dibenzo(a,h)anthracene with the Non-spatial and Watershed versions, due to its longer half-life in water, an emission of dibenzo(a,h)anthracene has a higher per kilogram ecotoxic impact according to Modified Watershed version, due to its higher aquatic ecotoxicity effect factor. These two substances have similar parameter values otherwise. This effect is particularly noticeable for metals, many of which have mid-range aquatic ecotoxicity effect factors but have elevated characterization factors as they do not degrade. A reduction in water residence time has a marked effect on the ecotoxic impact ranking of metal emissions.

5.1.2.1.2 Human Toxicity

Little difference exists between the human toxicity CF produced by the Non-Spatial model and the Watershed model, however a difference of one to three orders of magnitude exists between human toxicity CF produced by the Watershed version and the Modified Watershed version. The largest difference between the models was noted for n-nitrosodiphenylamine (factor 1.34E+03) and the smallest for acenaphthene (factor 6.91). For all three models, the largest and smallest CF were those of DDT and phenol, respectively (2.72E-02 & 5.33E-10 DALY/kg for the Modified Watershed model).

Table A6.1 of Appendix 6 lists the total oral and ingestion intake fractions per substance for each of the three models. These data are also presented below in Figures 5.4 and 5.5.

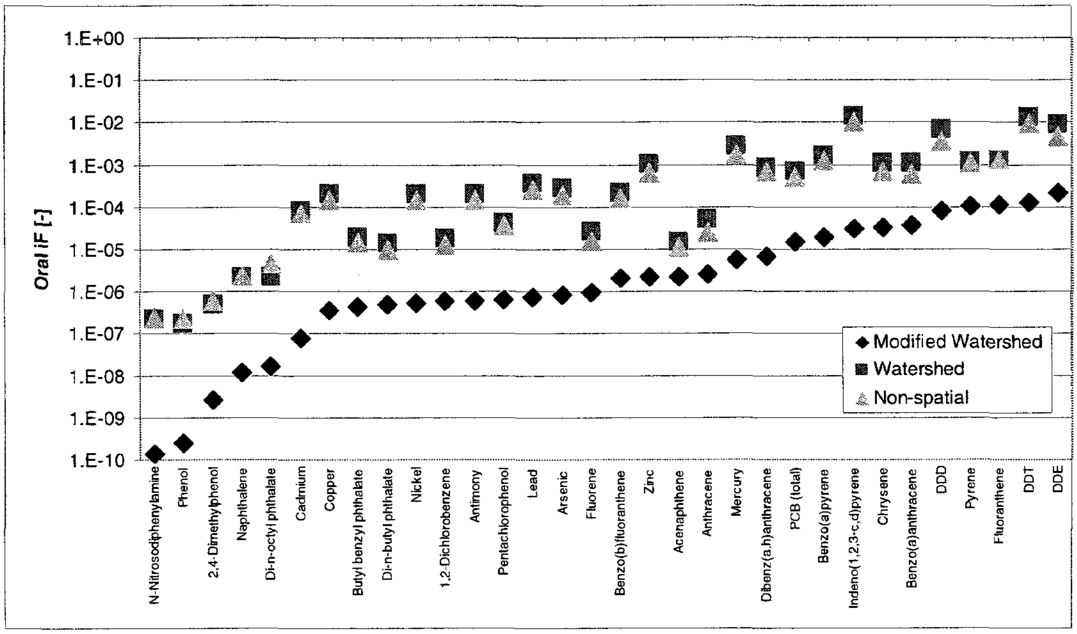


Figure 5.4. Oral intake fractions for an emission to water calculated using the Non-spatial, Watershed and Modified Watershed models.

Results indicated that with the exception of 1,2-dichlorobenzene, intake of the contaminants found at the MWW site are dominated by ingestion, with oral intake fractions between one and six orders of magnitude higher than those of inhalation intake fractions for organic substances and fourteen orders of magnitude higher for mercury.

Primary ingestion pathways include drinking water and fish consumption. Puget Sound is not used as a drinking water source (see Table A1.2 in Appendix 1) and this significantly reduces contaminant exposure. Additionally, the reduced residence time of Puget Sound compared to that of the other two models results in less contaminant being absorbed by food fish in the emission zone.

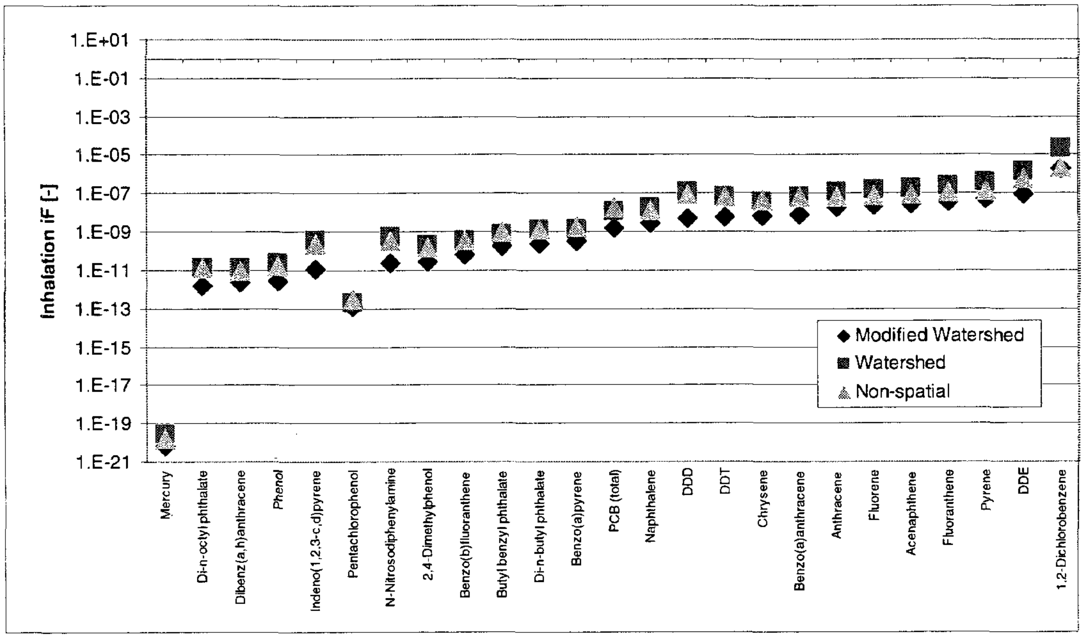


Figure 5.5. Inhalation intake fractions for an emission to water calculated using the Non-spatial, Watershed and Modified Watershed models.

Similar ranking effects due to residence time can be seen with the human toxicity characterization factors. Chrysene and indeno(1,2,3-cd)pyrene have the same toxicity ED10 values, however the latter is almost 10-times more persistent in water. As a result indeno(1,2,3-cd)pyrene has a higher toxicity CF with the Non-spatial and Watershed models, however a slightly lower CF with the Modified Watershed model, as the effect of this persistence is reduced by the lower water residence time.

5.1.2.2 Emissions to air

Figures 5.6 and 5.7 present the ecotoxicity and human toxicity characterization factors produced by the two IMPACT model versions for an emission to air. Table 5.2 below shows intake results as well as water, air and soil concentrations produced by both models for an emission of selected substances to air. Results for all substances can be found in Table A6.2 of Appendix 6.

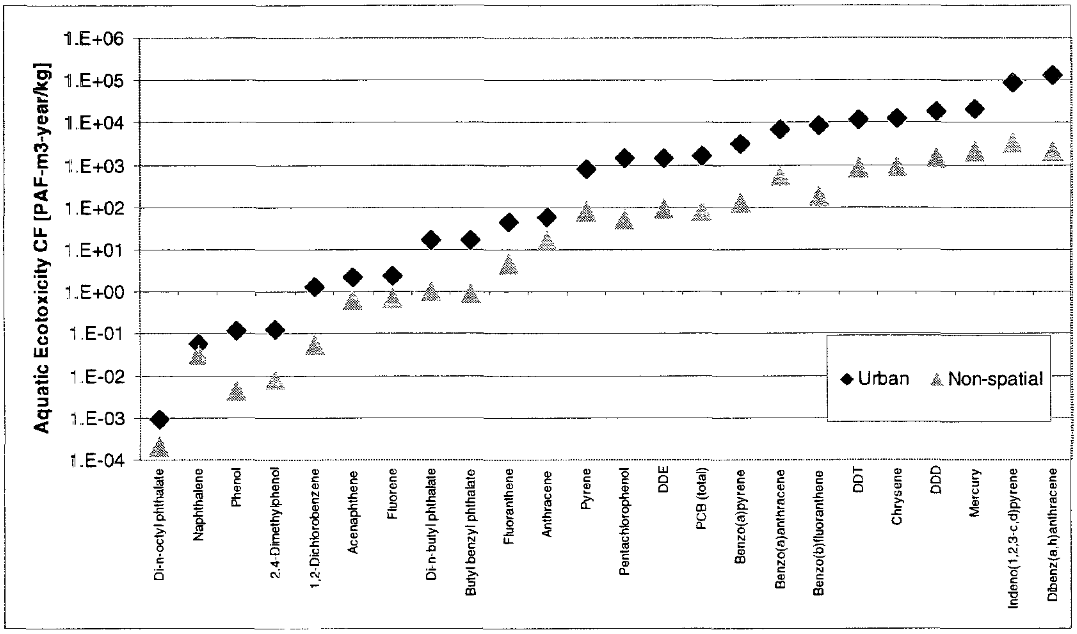


Figure 5.6. Ecotoxicity characterization factors for an emission to air evaluated by the Urban and Non-spatial IMPACT model versions.

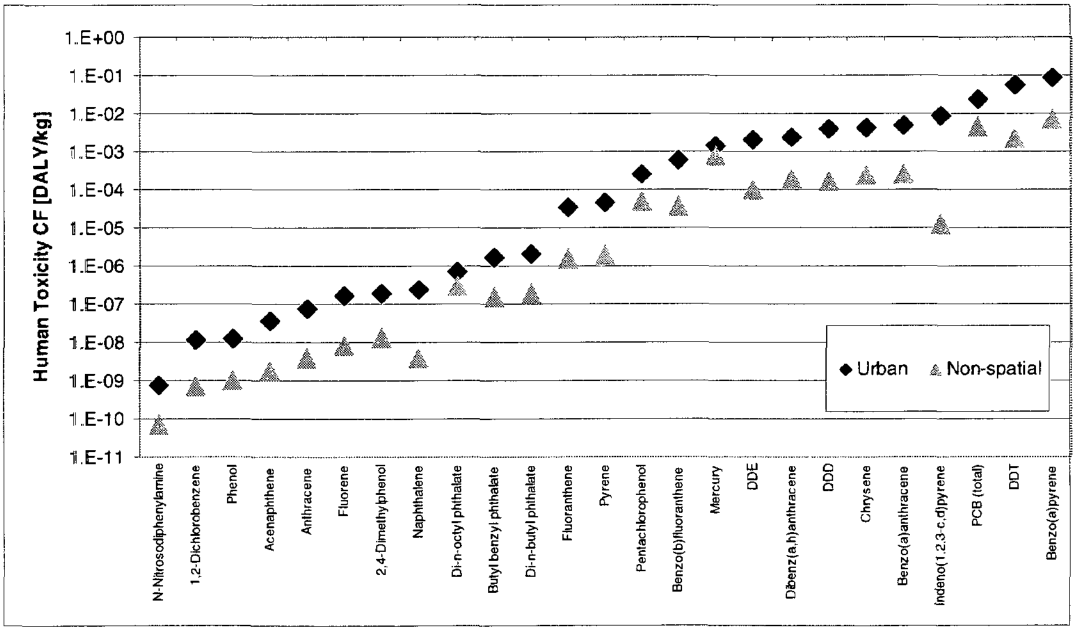


Figure 5.7. Human toxicity characterization factors for an emission to air evaluated by the Urban and Non-spatial IMPACT model versions.

5.1.2.2.1 Ecotoxicity

Ecotoxicity CF due to an emission to air within the Urban box of Tacoma were calculated to be 1.92 to 61.7 times higher than those for an emission to air within the Non-Spatial model. Water and soil concentrations evaluated with the Urban model for an emission in Tacoma, WA were consistently higher than those produced by the Non-spatial model for an emission in Canada. Water concentrations were approximately 20 to 514 times greater, and soil concentrations 9 to 455 times greater. Air concentrations with the Urban model were consistently lower by factors ranging from 3 to 25, with the exception of 1,2-dichlorobenzene which had slightly higher concentrations with the Urban model.

Several factors may contribute to the higher soil and water concentrations produced by the Urban model. The rainfall rate in the emission zone and in downwind zones of the Urban model is on average twice that of the Non-spatial model. Moreover, the soil-to-water surface area ratio for these zones is approximately 2.7 times that of the emission zone for the Non-spatial model (see Table A1.2 of Appendix 1). Each of these factors results in greater deposition to soil, as evidenced by the reduced air concentrations with the Urban model.

The higher rainfall rate in the immediate regions of the Urban box model emission zone will also result in higher deposition to water surfaces. The increased soil-to-water surface area ratio might otherwise result in less mass transfer to water were it not for the higher run-off rate in the area surrounding the emission zone of the Urban box model, exceeding that of the Non-spatial model zone by a factor of 5.4. Higher rainfall rate and an increased run-off rate from more highly contaminated soil gives the overall result of increased water concentrations and increased ecotoxicity CF due to air emissions modeled using the Urban box model.

The increased fraction of emissions entering the water has ranking effects on the substances, with indeno(1,2,3-cd)pyrene and dibenz(a,h)anthracene again switching first and second characterization factor rank. In this case, with a greater fraction of emissions reaching the water according to the Urban box model, the higher ecotoxicity of dibenz(a,h)anthracene results in a higher ecotoxicity characterization factor.

Table 5.2. Intake results, water, air and soil concentrations produced by Urban and Non-spatial models for an emission of selected substances to air. Complete results are available in Table A6.2 of Appendix 6.

		1,2-Dichlorobenzene	Acenaphthene	Anthracene	Benzo(a)pyrene
Non-spatial	Total Oral Intake Fraction	1.31E-07	3.24E-07	3.84E-06	6.92E-04
	Total Inhalation Intake Fraction	4.11E-06	3.14E-07	2.97E-07	1.70E-07
	Troposphere air concentration (kg/m^3)	3.02E-15	1.53E-15	1.52E-15	1.23E-15
	Water concentration (kg/m^3)	2.40E-14	6.47E-14	1.71E-13	3.89E-12
	Soil concentration in upper layers (kg/m^3)	1.41E-13	4.38E-13	1.71E-12	4.07E-10
	Drinking water (kg/yr)	4.08E-07	1.09E-06	2.84E-06	3.67E-05
	Unexposed produce direct (kg/yr)	1.52E-06	1.50E-05	7.97E-05	1.11E-03
	Exposed produce direct (kg/yr)	6.75E-05	9.56E-04	1.20E-02	2.27E+00
	Intake via eggs (kg/yr)	1.19E-09	3.98E-09	1.15E-07	6.32E-04
	Intake via milk (kg/yr)	4.38E-09	5.57E-08	2.61E-06	1.53E-02
	Intake via meat (kg/yr)	7.22E-08	6.52E-07	2.93E-05	1.72E-01
Urban	Total Oral Intake Fraction	2.21E-06	6.27E-06	7.37E-05	8.69E-03
	Total Inhalation Intake Fraction	2.71E-05	1.15E-06	1.06E-06	6.06E-07
	Troposphere air concentration (kg/m^3)	5.28E-15	2.59E-16	2.34E-16	1.02E-16
	Water concentration (kg/m^3)	9.68E-12	2.26E-12	5.84E-12	9.16E-10
	Soil concentration in upper layers (kg/m^3)	2.31E-11	1.36E-11	5.00E-11	1.47E-08
	Drinking water (kg/yr)	2.02E-04	8.99E-05	2.80E-04	6.38E-02
	Unexposed produce direct (kg/yr)	4.42E-04	6.48E-04	3.12E-03	1.51E-02
	Exposed produce direct (kg/yr)	1.83E-02	5.33E-02	6.11E-01	3.76E+01
	Intake via eggs (kg/yr)	7.03E-07	2.03E-07	5.87E-06	4.29E-03
	Intake via milk (kg/yr)	2.72E-05	3.47E-05	1.51E-03	1.17E+00
	Intake via meat (kg/yr)	2.16E-04	6.50E-04	2.80E-02	2.15E+01

5.1.2.2.2 Human Toxicity

Human toxicity CF due to an emission to air within the Urban box of Tacoma were calculated to be 1.81 to 702 times higher than those for an emission to air within the Non-Spatial model. As with emissions to water, results indicated that ingestion is the dominant exposure route for contaminants found at the MWW site emitted to air, with oral intake fractions approximately 2.2 to 1.40E+4 times greater than inhalation fractions for the Urban model and 3.4 to 6.00E+3 times greater for the Non-spatial model (see Table A7.2). Exceptions to this trend include 1,2-dichlorobenzene, naphthalene and n-nitrosodiphenylamine for which inhalation was found to be the dominant exposure route. For the Urban model, ingestion of phenol was approximately the same as inhalation. The same can be said of acenaphthene with the Non-spatial model.

Exposure to contaminants via ingestion was dominated by consumption of exposed produce, with the exception of mercury, for which drinking water was the dominant ingestion exposure pathway. Ingestion of contaminants via exposed produce was 5.5 to 270 times higher with the Urban box model than with the Non-spatial model. The yearly output of exposed produce per unit area is on average 2.3 times higher for the regions surrounding the Urban box emission zone than for that of the Non-spatial model. This difference combined with the increased deposition to soil noted above may account for much of the difference between the human toxicity CF calculated by the two models.

Meat consumption was a significant ingestion pathway for the substances with the highest K_{ow} values (see Table A6.2 for benzo(a)anthracene, benzo(a)pyrene, DDD, DDT, dibenz(a,h)anthracene, indeno(1,2,3-c,d)pyrene and PCB), demonstrating their high affinity for being sequestered in animal fat. Average meat production per unit area in the regions surrounding the Urban box emission zone is approximately twice that of the Non-spatial model zone, which combined with the factors mentioned above may account for the 11 to 360 factor difference in contaminant ingestion via meat between the two models.

5.2 Environmental assessment of two remediation options for the Middle Waterway contaminated site

5.2.1 Inventory results for Option 1 MNA

This section presents the principal LCI materials and inputs for Option 1: MNA.

5.2.1.1 Monitoring Program

The monitoring program of the MNA option resulted in the utilization of 2278.4 kg of diesel, primarily consumed by the boat used for sampling (2270 kg), with a small amount due to the 230.6 t-km of transport of samples to the laboratory (8.4 kg).

5.2.1.2 Contaminant emissions and time-integrated sediment concentration

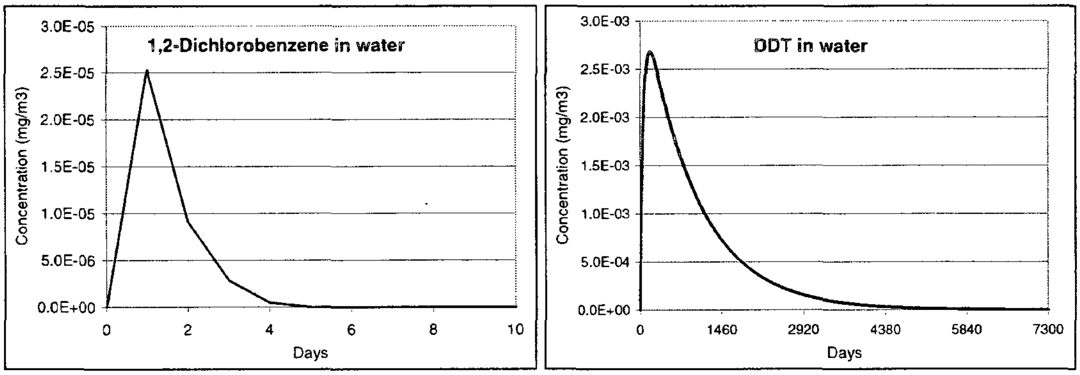
The inventory results associated with the contaminated site and off-site emissions were determined by the MNA mass balance model and are presented in this section. The output of the MNA model are expressions for contaminant concentrations as a function of time for the mixed sediments, deep sediments and water column. Examples of these expressions for the DDT concentration in water, **cw(t)**, mixed sediment, **cm(t)**, and deep sediment, **cds(t)**, are shown below in Equations 6.1, 6.2 and 6.3, respectively.

$$
\begin{aligned}
cw(t) = {} & 0.344e-2 * \exp(-0.106e-2*t) * \cos(0.532e-10*t) \\
& - 0.109e-5 * \exp(-0.106e-2*t) * \sin(0.532e-10*t) \\
& + 0.227e-7 * \exp(-0.243e-3*t) * \cos(0.483e-10*t) \\
& - 0.178e-10 * \exp(-0.243e-3*t) * \sin(0.483e-10*t) \\
& - 0.344e-2 * \exp(-0.170e-1*t) * \cos(0.132e-11*t) \\
& + 0.118e-7 * \exp(-0.170e-1*t) * \sin(0.132e-11*t)
\end{aligned}
\tag{Eq. 6.1}
$$

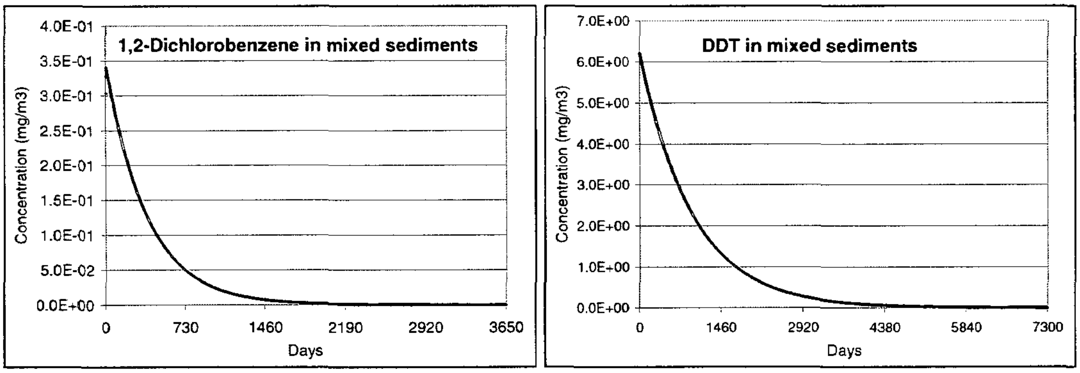
$$
\begin{aligned}
cm(t) = {} & 6.22*\exp(-0.106e-2*t) * \cos(0.532e-10*t) \\
& + 0.221e-7*\exp(-0.106e-2*t) * \sin(0.532e-10*t) \\
& + 0.229e-3*\exp(-0.170e-1*t) * \cos(0.132e-11*t) \\
& - 0.787e-9*\exp(-0.170e-1*t) * \sin(0.132e-11*t) \\
& + 0.432e-4*\exp(-0.243e-3*t) * \cos(0.483e-10*t) \\
& - 0.187e-7*\exp(-0.243e-3*t) * \sin(0.483e-10*t)
\end{aligned}
\tag{Eq. 6.2}
$$

$$
\begin{aligned}
cds(t) = {} & -3.40*\exp(-0.106e-2*t) * \cos(0.532e-10*t) \\
& + 0.453*\exp(-0.106e-2*t) * \sin(0.532e-10*t) \\
& + 19.4*\exp(-0.243e-3*t) * \cos(0.483e-10*t) \\
& - 0.840e-2*\exp(-0.243e-3*t) * \sin(0.483e-10*t) \\
& - 0.268e-4*\exp(-0.170e-1*t) * \cos(0.132e-11*t) \\
& - 0.188e-5*\exp(-0.170e-1*t) * \sin(0.132e-11*t)
\end{aligned}
\tag{Eq. 6.3}
$$

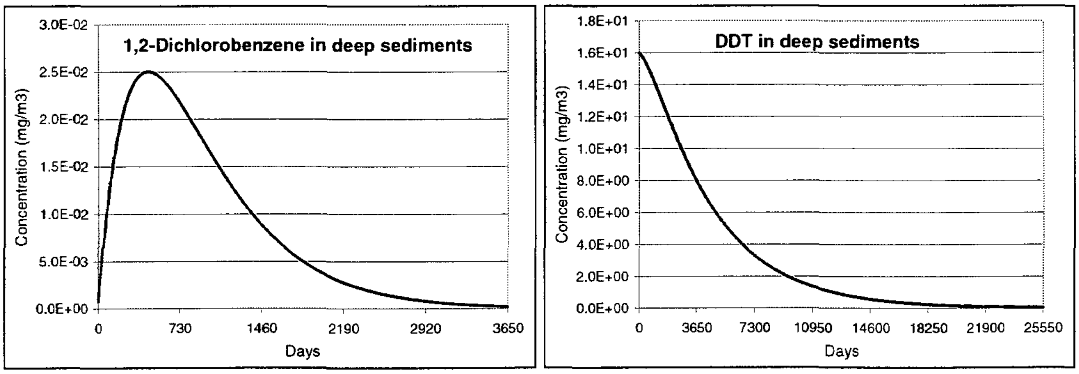
The expressions above can be used to generate concentration profiles with time that serve as a good first impression of the behaviour of each of the contaminants, e.g., whether it is persistent or is removed rapidly from the site via transport or transformation processes. The concentration profiles for 1,2-dichlorobenzene and DDT are shown below in Figures 5.8 to 5.13. Note that different time scales and concentration scales have been used in some cases to show the full progression of the contaminant profile.



Figures 5.8 and 5.9. Concentration profile estimate for 1,2-dichlorobenzene and DDT in the water column of the MWW contaminated site.



Figures 5.10 and 5.11. Concentration profile estimate for 1,2-dichlorobenzene and DDT in the mixed sediments of the MWW contaminated site.



Figures 5.12 and 5.13. Concentration profile estimate for 1,2-dichlorobenzene and DDT in the deep sediments of the MWW contaminated site.

As the presentation of concentration profiles for each contaminant would be unwieldy, the water column and mixed sediment concentration curves will be described by their peak concentration and the amount of time to achieve a ten-fold decrease in concentration following this peak (Table 5.3 and Table 5.5). Where available, Washington State water quality criteria and sediment quality objectives (SQO) for the CBNT Superfund site are included for comparison, as well as the time in excess of the criteria. The lowest detection limit reported by Anchor & Foster Wheeler (2001) for contaminants in water is also included as a benchmark for comparison, to indicate if the predictions made by the model are within an unverifiable range. It should be noted that the sediment concentrations reported here are predicted averages for the study site. Individual sampling stations may have much higher or lower concentrations. The lowest detection limit reported by Anchor & Foster Wheeler (2001) for contaminants in sediment is included for comparison.

Table 5.3. Descriptors of contaminant concentration profiles predicted by the MNA model for the MWW water column: peak concentration [mg/m^3], lowest reported detection limit [mg/m^3] and the time required to reach a 10-fold reduction in concentration following the peak [days]. Shaded cells exceeded at least one (acute or chronic) marine surface water quality criterion.

Substance	**Peak concentration**	**Detection Limit**	**10-fold conc. reduction [d]**
1,2-Dichlorobenzene	2.53E-05	2.00E+00	3
2,4-Dimethylphenol	7.23E+02	9.00E+00	2
Acenaphthene	1.37E+02	1.80E+00	5
Anthracene	7.68E+01	9.00E+00	2
Benzo(a)anthracene	8.47E+00	9.00E+00	3
Benzo(a)pyrene	1.28E+00	9.00E+00	10
Benzo(b)fluoranthene	6.23E+00	9.00E+00	94
Butylbenzylphthalate	3.11E-01	9.00E+00	9
Chrysene	1.57E+01	9.00E+00	3
DDD	9.21E-04	9.40E-03	1743
DDE	7.14E-02	9.40E-03	10
DDT	2.68E-03	9.40E-03	226
Dibenzo(a,h)anthracene	6.50E-02	9.00E+00	823
Di-n-butylphthalate	3.28E+01	9.00E+00	3
Di-n-octylphthalate	3.20E-06	9.00E+00	1716
Fluoranthene	1.81E+02	1.20E+00	5
Fluorene	9.85E+01	9.00E+00	8
Indeno(1,2,3-cd)pyrene	1.19E-01	9.00E+00	4160
Naphthalene	1.48E+03	2.00E+00	3
N-Nitrosodiphenylamine	7.45E+00	3.00E+00	3
PCB (total)	2.64E-01	9.40E-02	3027
Pentachlorophenol	6.22E+01	1.80E+00	2
Phenol	1.97E+04	9.00E+00	2
Pyrene	1.08E+02	9.00E+00	3
Antimony	6.08E+04	5.00E+00	3
Arsenic	1.92E+03	5.00E+00	39
Cadmium	4.14E-02	5.00E-02	18263
Copper	1.52E+05	2.00E+00	7
Lead	2.46E+01	1.00E+00	18263
Mercury	2.82E+01	2.00E-01	147
Nickel	4.44E+04	5.00E+00	3
Zinc	5.73E+05	7.00E+00	4

According to the MNA model, thirty of the thirty-two substances exceeded the detection limit in water. The following eleven substances did not exceed the detection limit:

- 1,2-dichlorobenzene
- benzo(a)anthracene
- benzo(a)pyrene
- benzo(b)fluoranthene
- butylbenzylphthalate
- DDD
- DDT
- dibenzo(a,h)anthracene
- di-n-octylphthalate
- indeno(1,2,3-cd)pyrene
- cadmium

Table 5.4 provides the values for available marine surface water quality criteria for contaminants present in MWW and the number of days that the MNA model predicted the water column would exceed these criteria. DDE, pentachlorophenol and total PCBs, as well as all metals except for antimony and cadmium exceeded either the acute or the chronic marine surface water quality criteria. Washington State does not have a surface water quality criterion for antimony. A contaminant was considered to have exceeded the chronic water quality criterion for one day if the four-day average up to and including that day were in excess of the criterion value. Three days were then added to each chemical in exceedance of the criterion for one or more days.

Table 5.4. Marine Surface Water Quality Criteria for Washington State (Ecology, 2006). Units are in mg/m^3 for comparison to MNA model output.

Substance	**Acute** [a]	**Days exceeded**	**Chronic** [b]	**Days exceeded**
Arsenic	69.0	58	36.0	74
Cadmium	42.0	-	9.3	-
Copper	4.8	1535	3.1	2071
DDE	0.13	1	0.001	-
Lead	210.0	-	8.1	36031
Mercury	1.8	205	0.025	5097
Nickel	74.0	9	8.2	14
Pentachlorophenol	13.0	1	7.9	4
PCB [c]	10.0	-	0.030	2883
Zinc	90.0	12	81.0	15

a: 24-hour average concentration not to be exceeded

b: A 4-day average concentration not to be exceeded more than once every three years on the average

c: Acute and chronic water quality criteria for PCB are both a 24-hour average concentration not to be exceeded

Table 5.5. Descriptors and comparison benchmarks of contaminant concentration profiles predicted by the MNA model for the MWW mixed sediment: sediment quality objectives (SQO) [mg/m^3], peak concentration [mg/m^3], lowest reported detection limit [mg/m^3] and the time required to reach a 10-fold reduction in concentration following the peak [days]. The shaded cell indicates that the peak concentration exceeded the SQO.

Substance	**SQO**	**Peak concentration**	**Detection limit**	**10-fold conc. reduction [d]**
1,2-Dichlorobenzene	1.32E+02	3.23E-01	6.05E+00	868
2,4-Dimethylphenol	7.63E+01	1.93E+01	4.21E+01	75
2-Methylnaphthalene	1.76E+03	3.93E+02	1.03E+02	388
Acenaphthene	1.32E+03	4.32E+02	6.05E+01	21
Acenaphthylene	3.42E+03	1.07E+03	1.42E+02	1481
Anthracene	2.52E+03	2.44E+03	8.68E+01	1754
Benzo(a)anthracene	4.21E+03	2.00E+03	6.84E+01	1630
Benzo(a)pyrene	4.21E+03	3.58E+03	9.73E+01	1749
Butylbenzylphthalate	2.37E+03	8.32E+00	1.29E+02	272
Chrysene	7.36E+03	2.67E+03	9.47E+01	2024
DDD	4.21E+01	3.74E-01	3.95E+00	2172
DDE	2.37E+01	3.87E+00	3.95E+00	2163
DDT	8.94E+01	6.22E+00	3.95E+00	2165
Dibenzo(a,h)anthracene	6.05E+02	8.29E+01	6.84E+01	1973
Di-n-butylphthalate	3.68E+03	3.26E+01	1.05E+02	40
Di-n-octylphthalate	1.63E+04	1.74E-02	3.42E+01	593
Fluoranthene	6.58E+03	5.63E+03	1.16E+02	1485
Fluorene	1.42E+03	6.03E+02	7.36E+01	390
Indeno(1,2,3-cd)pyrene	1.81E+03	6.29E+02	7.10E+01	1898
Naphthalene	5.52E+03	1.00E+03	9.73E+01	374
N-Nitrosodiphenylamine	7.36E+01	5.67E+00	1.53E+01	600
PCB (Total)	7.89E+02	2.09E+02	9.99E+01	2641
Pentachlorophenol	9.47E+02	3.56E+01	4.21E+01	601
Phenol	1.10E+03	6.36E+02	3.16E+01	75
Pyrene	8.68E+03	4.79E+03	1.71E+02	2305
Antimony	3.95E+05	1.97E+04	6.58E+03	2935
Arsenic	1.50E+05	7.85E+04	6.58E+02	2807
Cadmium	1.34E+04	6.49E+02	6.58E+02	2877
Copper	1.03E+06	7.06E+05	1.32E+03	2812
Lead	1.18E+06	3.86E+05	3.95E+03	2877
Mercury	1.55E+03	3.62E+03	1.05E+02	2804
Nickel	3.68E+05	4.05E+04	1.32E+03	2819
Zinc	1.08E+06	5.92E+05	1.32E+03	2817

With the exception of mercury, no SQO was exceeded by the peak average mixed sediment concentration predicted by the MNA model. It should be noted that the average concentration was of the same order of magnitude as the SQO for twenty-five of the thirty-two chemicals, indicative of the high concentrations found at individual sampling stations.

Average concentrations of eleven out of the thirty-two contaminants were predicted to not exceed the detection limit in the mixed sediments:

- 1, 2-dichlorobenzene
- DDD
- di-n-octylphthalate
- cadmium
- 2,4-dimethylphenol
- DDE
- n-nitrosodiphenylamine
- lead
- butylbenzylphthalate
- di-n-butylphthalate
- pentachlorophenol

The MNA model predicted that the following eight substances would not exceed the detection limit in the deep sediments:

- 1,2-dichlorobenzene
- di-n-butylphthalate
- 2,4-dimethylphenol
- n-nitrosodiphenylamine
- butylbenzylphthalate
- pentachlorophenol

Of the thirty-two substances evaluated, ten had average concentrations in the deep sediment that were predicted to exceed the CBNT SQO:

- anthracene
- benzo(a)anthracene
- benzo(a)pyrene
- acenaphthene
- fluoranthene
- fluorene
- mercury
- pyrene
- copper
- chrysene

All other contaminants were within one order of magnitude of the SQO with the exception of 1,2-dichlorobenzene, di-n-butylphthalate, n-nitrosodiphenylamine and pentachlorophenol. Contaminants were predicted to persist much longer in the deep sediments than in the mixed sediments. Notably, average metal concentrations were predicted to remain at levels above or within an order of magnitude of SQO for the full

100 years evaluated. DDT and its derivatives, as well as several PAHs (acenaphthylene, anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene and indeno(1,2,3-cd)pyrene) showed significant persistence in the deep sediments, taking 10 years or more to achieve a 10-fold decrease in the predicted average concentration.

Table 5.6. Descriptors and comparison benchmarks of contaminant concentration profiles predicted by the MNA model for the MWW deep sediment: sediment quality objectives (SQO) [mg/m^3], peak concentration [mg/m^3], lowest reported detection limit [mg/m^3] and the time required to reach a 10-fold reduction in concentration following the peak [days]. Shaded cells indicate that the SQO was exceeded by the peak concentration.

Substance	**SQO**	**Peak concentration**	**Detection limit**	**10-fold conc. reduction [d]**
1,2-Dichlorobenzene	1.32E+02	2.68E-02	6.05E+00	1804
2,4-Dimethylphenol	7.63E+01	3.66E+01	4.21E+01	77
Acenaphthene	1.32E+03	1.55E+03	6.05E+01	21
Acenaphthylene	3.42E+03	2.40E+03	1.42E+02	3308
Anthracene	2.52E+03	8.53E+03	8.68E+01	3537
Benzo(a)anthracene	4.21E+03	1.01E+04	6.84E+01	2471
Benzo(a)pyrene	4.21E+03	5.29E+03	9.73E+01	3780
Butylbenzylphthalate	2.37E+03	5.03E-01	1.29E+02	479
Chrysene	7.36E+03	8.65E+03	9.47E+01	3780
DDD	4.21E+01	7.68E+00	3.95E+00	3780
DDE	2.37E+01	6.65E+00	3.95E+00	3780
DDT	8.94E+01	1.60E+01	3.95E+00	3780
Dibenzo(a,h)anthracene	6.05E+02	3.26E+02	6.84E+01	3780
Di-n-butylphthalate	3.68E+03	6.91E-01	1.05E+02	54
Di-n-octylphthalate	1.63E+04	3.40E+03	3.42E+01	64
Fluoranthene	6.58E+03	1.01E+04	1.16E+02	3476
Fluorene	1.42E+03	2.49E+03	7.36E+01	460
Indeno(1,2,3-cd)pyrene	1.81E+03	1.08E+03	7.10E+01	3780
Naphthalene	5.52E+03	1.70E+03	9.73E+01	449
N-Nitrosodiphenylamine	7.36E+01	3.27E-01	1.53E+01	1151
PCB (Total)	7.89E+02	3.06E+02	9.99E+01	1816
Pentachlorophenol	9.47E+02	4.77E+00	4.21E+01	1152
Phenol	1.10E+03	1.71E+02	3.16E+01	82
Pyrene	8.68E+03	9.73E+03	1.71E+02	13298
Antimony	3.95E+05	8.21E+04	6.58E+03	30788
Arsenic	1.50E+05	1.14E+05	6.58E+02	23812
Cadmium	1.34E+04	1.44E+03	6.58E+02	*
Copper	1.03E+06	1.29E+06	1.32E+03	26871
Lead	1.18E+06	5.98E+05	3.95E+03	*
Mercury	1.55E+03	6.23E+03	1.05E+02	21803
Nickel	3.68E+05	5.33E+04	1.32E+03	28631
Zinc	1.08E+06	7.91E+05	1.32E+03	28399

*Concentration did not show a reduction over the 100-year time period evaluated.

5.2.1.3 LCI quantities for MNA primary impact assessment

The values calculated for mass emitted via air and water as well as the time-integrated mixed sediment concentration for the thirty-two contaminants are provided in Table 5.7. For the perspective considering the contaminated site as part of the ecosphere, the time-integrated sediment concentrations at the MWW site are included in the LCI of the MNA management option. For the perspective considering the contaminated site as part of the technosphere, these quantities are excluded from the LCI. Both perspectives include off-site emissions as part of the LCI.

Table 5.7. Predicted mass emitted via water and air [kg] from MWW site to Commencement Bay and Puget Sound and time-integrated mixed sediment contaminant concentration[kg-year/m^3]

	Water emissions		Air emissions		Time-integrated sediment concentration	
	50 years	100 years	50 years	100 years	50 years	100 years
Substance						
1,2-Dichlorobenzene	1.93E-01	1.93E-01	2.35E-01	2.35E-01	1.22E-04	1.22E-04
2,4-Dimethylphenol	9.65E+01	9.65E+01	1.08E+02	1.08E+02	6.61E-04	6.61E-04
Acenaphthene	1.24E+01	1.24E+01	3.10E+01	3.10E+01	3.92E-03	3.92E-03
Anthracene	1.70E+00	1.70E+00	1.22E+01	1.22E+01	6.88E-01	6.88E-01
Benzo(a)anthracene	3.19E-02	3.19E-02	8.68E-04	8.68E-04	1.86E+00	1.86E+00
Benzo(a)pyrene	1.43E-01	1.43E-01	4.88E-04	4.88E-04	1.42E+00	1.42E+00
Benzo(b)fluoranthene	8.19E-01	8.19E-01	3.50E-02	3.50E-02	2.72E+00	2.72E+00
Butylbenzylphthalate	3.38E-02	3.38E-02	2.16E-04	2.16E-04	9.82E-04	9.82E-04
Chrysene	6.78E-02	6.78E-02	4.11E-04	4.11E-04	2.35E+00	2.35E+00
DDD	1.31E-03	1.31E-03	1.91E-05	1.91E-05	3.53E-04	3.53E-04
DDE	9.41E-04	9.41E-04	2.34E-02	2.34E-02	3.64E-03	3.64E-03
DDT	1.95E-02	1.95E-02	2.52E-03	2.52E-03	5.85E-03	5.85E-03
Dibenzo(a,h)anthracene	9.60E-03	9.60E-03	1.25E-06	1.25E-06	7.10E-02	7.10E-02
Di-n-butylphthalate	4.17E+00	4.17E+00	6.45E+00	6.45E+00	5.62E-04	5.62E-04
Di-n-octylphthalate	2.63E-03	2.63E-03	1.02E-05	1.02E-05	6.81E-05	6.81E-05
Fluoranthene	1.04E+01	1.04E+01	6.92E-01	6.92E-01	3.63E+00	3.63E+00
Fluorene	7.00E+00	7.00E+00	2.65E+01	2.65E+01	1.02E-01	1.02E-01
Indeno(1,2,3-cd)pyrene	1.67E-01	1.67E-01	4.37E-05	4.37E-05	5.18E-01	5.18E-01
Naphthalene	2.02E+02	2.02E+02	2.84E+02	2.84E+02	1.64E-01	1.64E-01
N-Nitrosodiphenylamine	1.02E+00	1.02E+00	1.47E+00	1.47E+00	1.48E-03	1.48E-03
PCB (total)	4.08E-01	4.08E-01	4.85E-02	4.85E-02	2.40E-01	2.40E-01
Pentachlorophenol	9.02E+00	9.02E+00	8.69E-04	8.69E-04	9.31E-03	9.31E-03
Phenol	3.17E+03	3.17E+03	9.74E+00	9.74E+00	2.18E-02	2.18E-02
Pyrene	1.99E+00	1.99E+00	1.48E-01	1.48E-01	4.79E+00	4.79E+00
Antimony	2.08E+04	2.08E+04	0.00E+00	0.00E+00	2.83E+01	3.28E+01
Arsenic	9.44E+02	9.44E+02	0.00E+00	0.00E+00	9.58E+01	9.58E+01
Cadmium	2.04E+00	2.19E+00	0.00E+00	0.00E+00	8.31E-01	8.34E-01
Copper	5.44E+04	5.44E+04	0.00E+00	0.00E+00	8.71E+02	8.78E+02
Lead	1.22E+03	1.30E+03	0.00E+00	0.00E+00	4.94E+02	4.96E+02
Mercury	1.42E+01	1.42E+01	8.71E+00	8.71E+00	4.40E+00	4.40E+00
Nickel	1.53E+04	1.53E+04	0.00E+00	0.00E+00	5.10E+01	5.26E+01
Zinc	1.98E+05	1.98E+05	0.00E+00	0.00E+00	7.41E+02	7.61E+02

5.2.2 Inventory results for Option 2: Dredging and confined disposal

This section presents the principal LCI materials and inputs for the Remediation Activities. The principal construction materials required for dredging and confined disposal are indicated in Table 5.8, totalling a mass of 2.84E+4 tons, dominated by 1.61E+4 tons of backfill required to bring the dredged waterway back to grade.

Table 5.8. Principal construction materials required for dredging and confined disposal.

Materials	Mass[t]
Gravel	2.02E+04
Clay	8.17E+03
PVC piping	3.04E-01
HDPE liner	1.36E+01

Diesel was an important inventory input for the LCI of the Remediation Activities. The total mass of diesel consumed (2.68E+5) was greater than the combined mass of construction materials used. Table 5.9 below outlines the diesel consumption required for Option 2, with the associated transport quantity [t-km] or energy use [HP-h] indicated as well.

Table 5.9. Diesel consumption for dredging and confined disposal, with associated transport quantity or energy use.

Activity	**Diesel consumption [kg]**	**Energy Use [HP-h]**
Dredging - derrick	1.93E+04	1.06E+05
Dredging - suction dredge	1.57E+03	6.58E+03
Backfill placement - loader	4.21E+03	2.43E+04
Ground clearing - dozer	4.00E+03	2.40E+04
Piling - dozer	9.99E+02	6.00E+03
Profiling - excavator	6.22E+02	3.30E+03
Compacting - roller	2.54E+03	1.53E+04
Sampling - boat	1.14E+02	500
		Transport [t-km]
Sediment transport	1.83E+05	5.02E+06
Backfill transport	2.93E+04	8.04E+05
Clay transport	1.49E+04	4.08E+05
PVC piping transport	1.67E-01	4.57E+00
Liner transport	2.47E+01	6.78E+02
Gravel transport	7.45E+03	2.04E+05
Sample transport	4.91E-01	13.5

Diesel consumption was dominated by sediment transport over the assumed 40 km to the CDF, which represented the largest fraction of the 6.44E+6 t-km of transport associated with the Remediation Activities. Dredging using the derrick consumed the greatest quantity of diesel among the non-transport related activities, accounting for 57% of the 1.86E+5 HP-h of non-transport related energy use.

5.3 LCIA

5.3.1 Impact assessment of Option 1: MNA

5.3.1.1 Impacts of the MNA monitoring program

Figure 5.14 shows the relative impact of the MNA monitoring program components, sample collection and sample transport, towards IMPACT 2002+ damage impact categories. It is apparent that the bulk of the monitoring program impacts are caused by the sample collection, due to the diesel consumption of the boat used to collect samples.

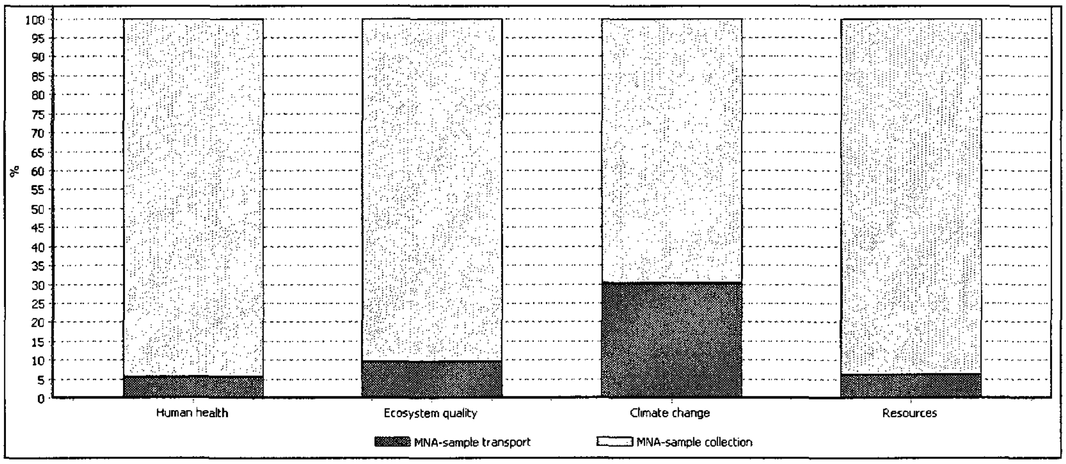


Figure 5.15. Relative contribution of sample collection and sample transport as part of the MWW site MNA management option to IMPACT 2002+ damage impact categories.

5.3.1.2 On-site sediment ecotoxicological impacts

The on-site ecotoxicological impacts associated with contamination in the sediments, calculated using the time-integrated average sediment concentrations and sediment ecotoxicity effect factors are shown in Figure 5.16. On-site ecotoxicological impacts for all chemicals studied over the 100-year time period totalled 8.86E+08 PAF-m^3-year. The four substances that caused the greatest ecotoxicity impacts on-site were all metals (nickel, antimony, zinc and copper), with 96% of the on-site impact being caused by zinc and copper alone (2.55E+08 and 5.98E+08 PAF-m^3-year, respectively). The remaining metals all scored high ecotoxicity impacts relative to the organic substances, with the exception of cadmium (2.97 PAF-m^3-year). Of the six organic substances causing the greatest on-site impacts, all were high molecular weight PAH (four rings or greater), with the exception of anthracene (three-rings). Chrysene was responsible for the greatest on-site ecotoxicity impact of the organic substances (8.09E+05 PAF-m^3-year) and di-n-octylphthalate caused the least (2.21E-07 PAF-m^3-year).

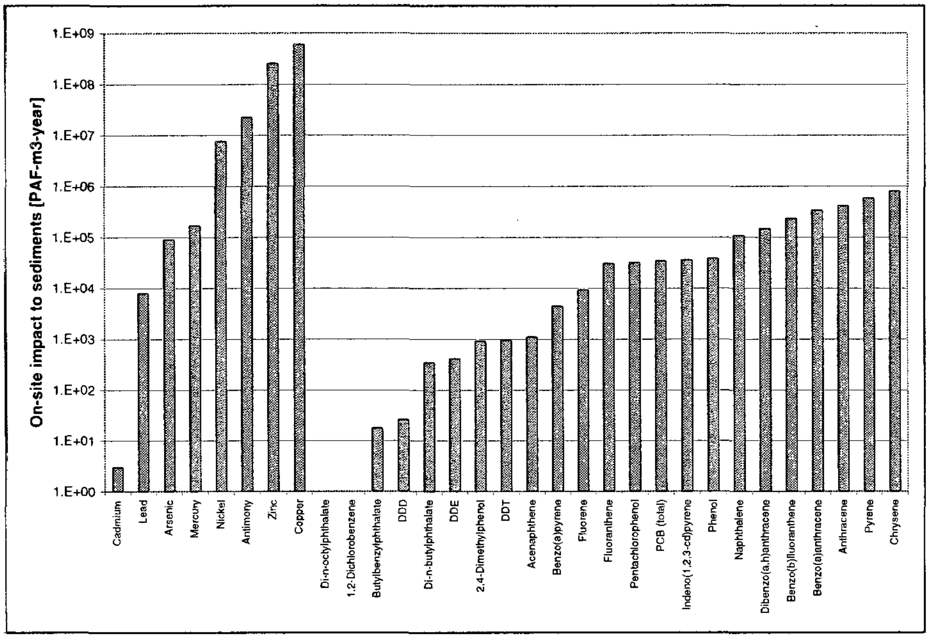


Figure 5.16. On-site ecotoxicity impacts to sediments at the MWW site.

5.3.1.3 Off-site ecotoxicity impacts due to water emissions from MWW site

The off-site ecotoxicity impacts associated with MNA water emissions, as evaluated by the three model versions are presented in Figures 5.17, 5.18 and 5.19. Off-site ecotoxicological impacts for all water emissions over the 100-year time period totalled 1.17E+09, 1.55E+09 and 3.66E+06 PAF-m^3-year for the Non-spatial, Watershed and Modified Watershed models, respectively. Metals were responsible for the bulk of the off-site ecotoxicological impact, with 96% being attributed to zinc and copper alone.

Of the organic substances, indeno(1,2,3-cd)pyrene, naphthalene, pyrene and benzo(b)fluoranthene caused the greatest off-site impacts, with indeno(1,2,3-cd)pyrene causing 42% and 44% of the organic substance off-site impact for the Non-spatial and Watershed evaluations, respectively. The ranking of organic substances changed with the Modified Watershed model and naphthalene was found to cause the greatest off-site impact, accounting for 27% of the 2.43E+03 PAF-m^3-year impact off-site.

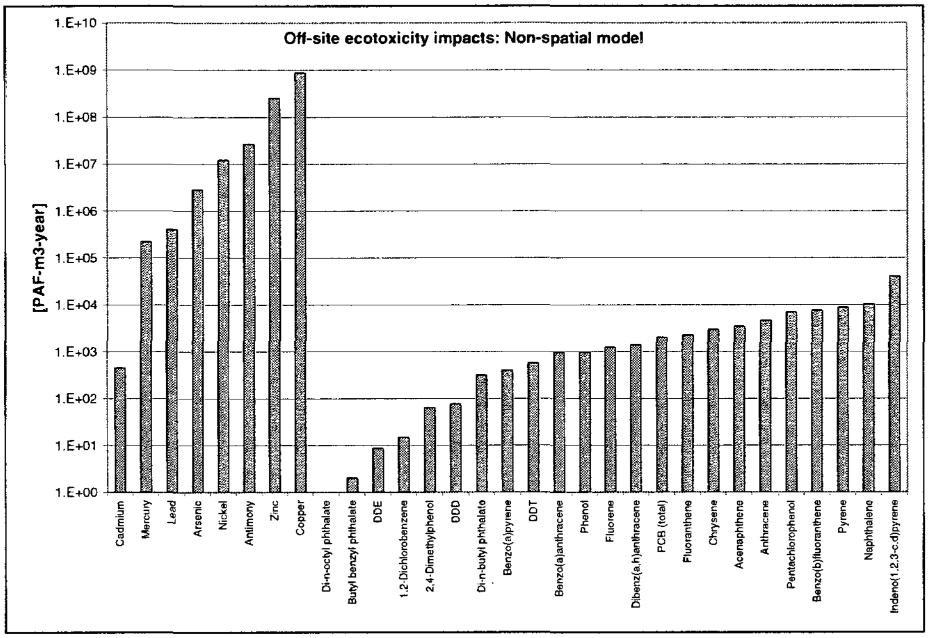


Figure 5.17. Off-site ecotoxicity impacts as evaluated by the Non-spatial model.

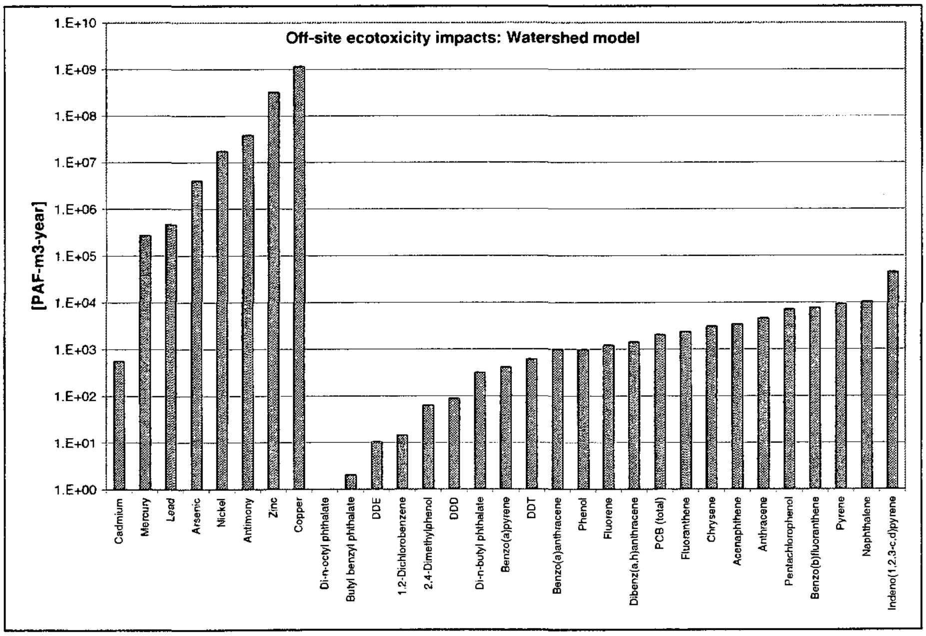


Figure5.18. Off-site ecotoxicity impacts as evaluated by the Watershed model.

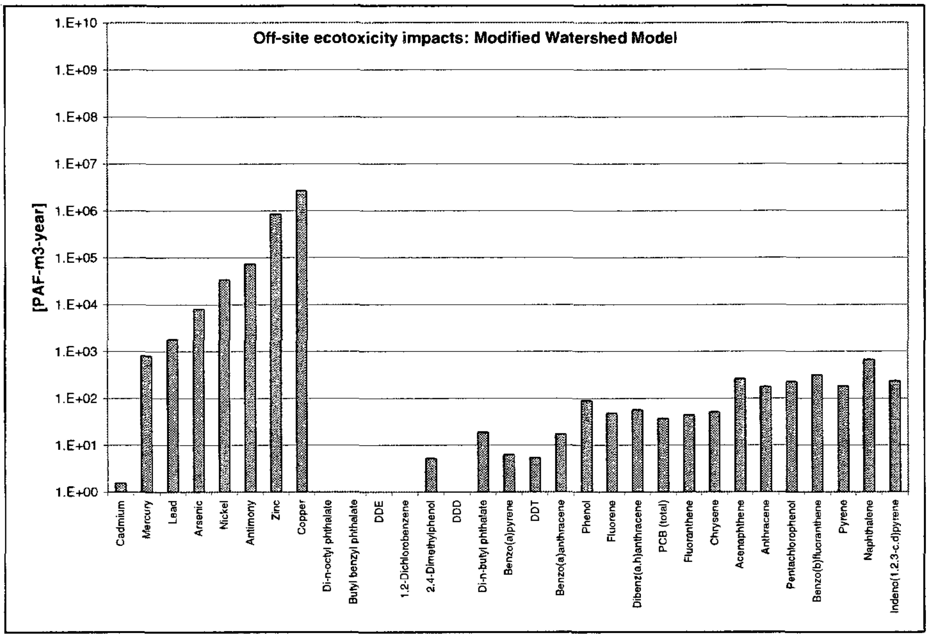


Figure 5.19. Off-site ecotoxicity impacts as evaluated by the Modified Watershed model.

5.3.1.4 Off-site ecotoxicity impacts due to air emissions from MWW site

The off-site ecotoxicity impacts associated with MNA air emissions, as evaluated by the Urban box and Non-spatial model versions are presented in Figure 5.20. Off-site ecotoxicological impacts for all air emissions over the 100-year time period totalled 1.91E+05 PAF-m^3-year for the Urban box model, approximately 5% of the impact due to water emissions evaluated by the Modified Watershed model and 0.01% of water emission impacts evaluated by the Watershed model. Air emissions evaluated by the Non-spatial model totalled 1.85E+04 PAF-m^3-year, approximately 0.002% of water emission ecotoxic impacts as evaluated by the Non-spatial model. Mercury was responsible for 99% of the ecotoxic impacts due to air emissions for both the Urban box and Non-spatial models.

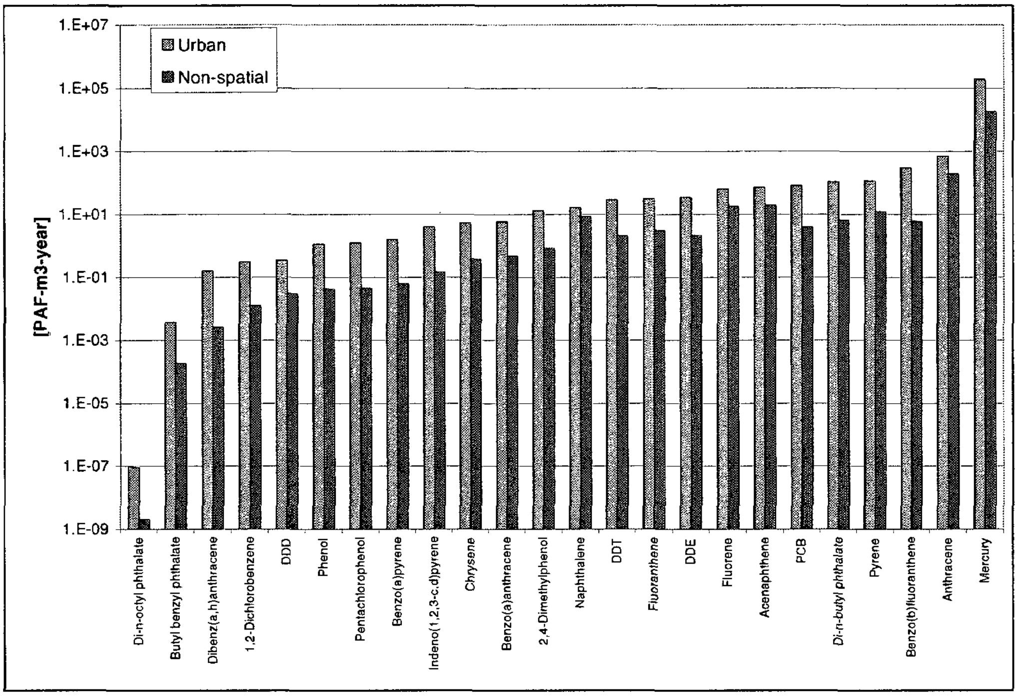


Figure 5.20. Off-site ecotoxicity impacts associated with MNA air emissions, as evaluated by the Urban box and Non-spatial IMPACT model versions.

5.3.1.5 Off-site human toxicity impacts due to water emissions from MWW site

The off-site human toxicity impacts associated with MNA water emissions, as evaluated by the three model versions are presented in Figures 5.21, 5.22 and 5.23. Off-site toxicity impacts for all chemicals studied over the 100-year time period totalled 6.76E+01, 1.02E+02, and 2.29E-01 DALY for the Non-spatial, Watershed and Modified Watershed models, respectively. Metals were responsible for the bulk of the off-site ecotoxicological impact, with 77 to 82% being attributed to zinc alone depending on the model.

For the Non-spatial and Watershed models, 92% of the off-site toxicity impacts due to organic emissions to water were caused by four chemicals: indeno(1,2,3-c,d)pyrene, benzo(a)pyrene, PCB and DDT. With the Modified Watershed model these substances made up 79% of this impact.

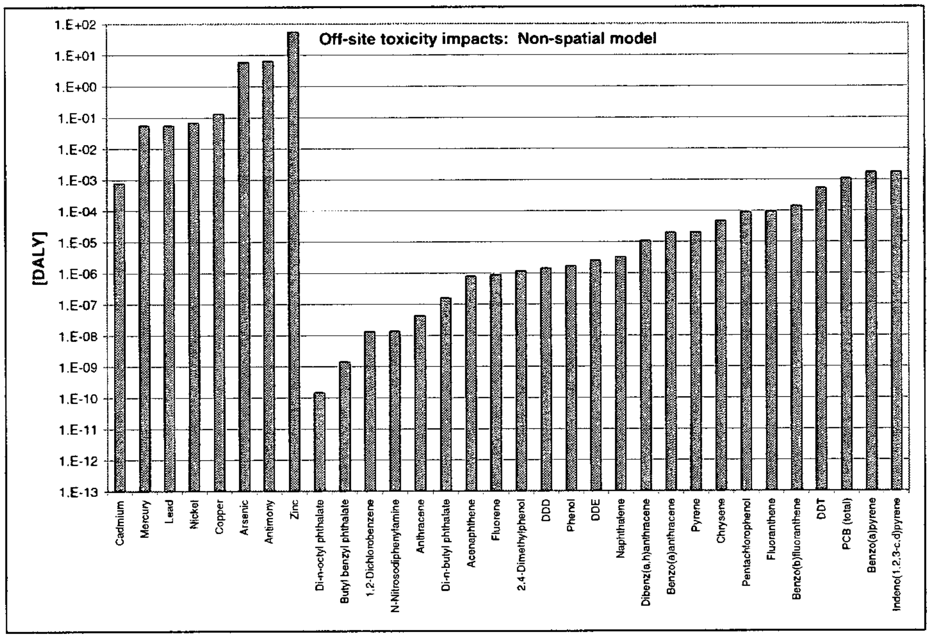


Figure 5.21. Off-site toxicity impacts as evaluated by the Non-spatial model.

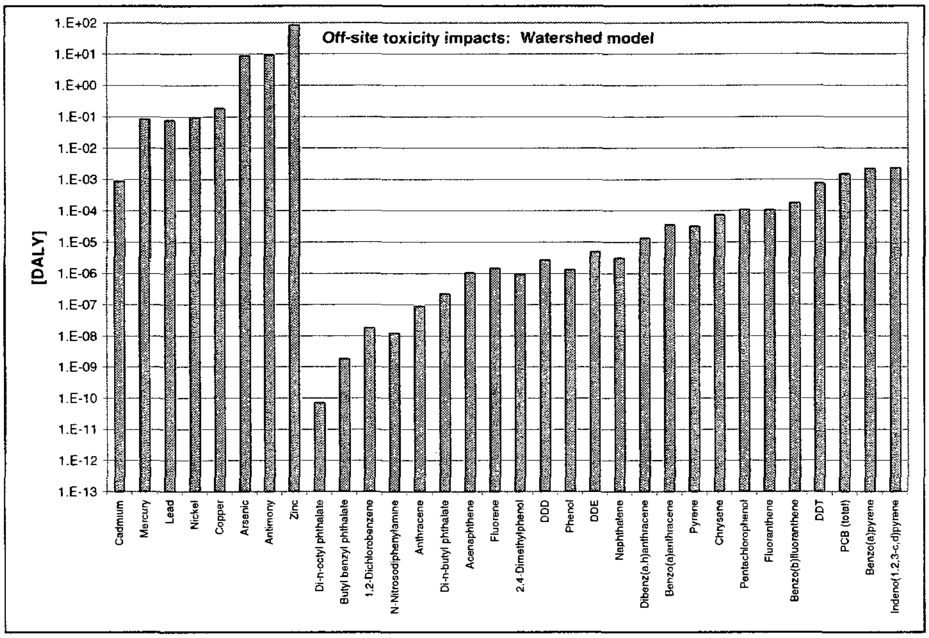


Figure 5.22. Off-site toxicity impacts as evaluated by the Watershed model.

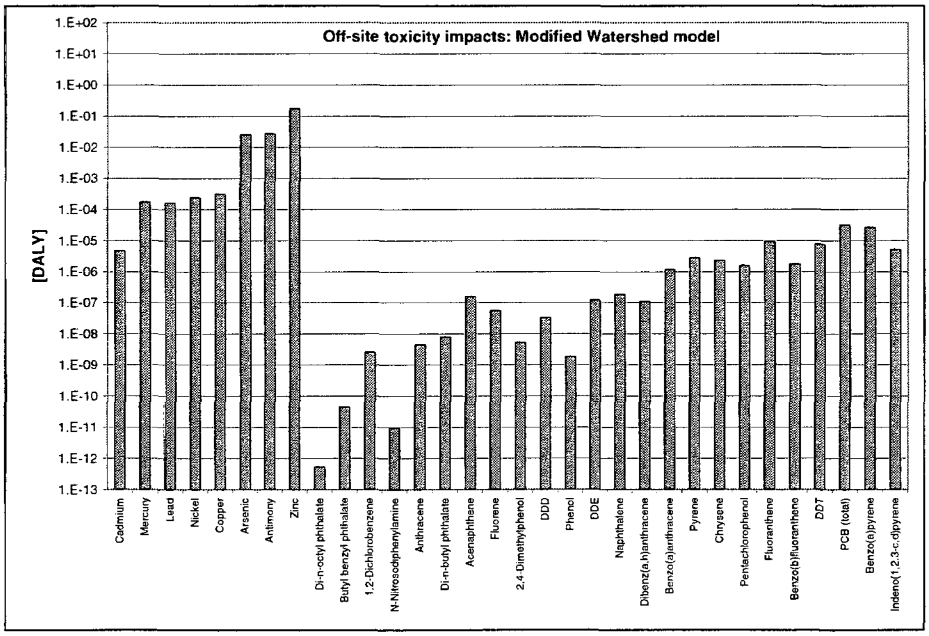


Figure 5.23. Off-site toxicity impacts as evaluated by the Modified Watershed model.

5.3.1.6 Off-site human toxicity impacts due to air emissions from MWW site

The off-site human toxicity impacts associated with MNA air emissions, as evaluated by the Urban box and Non-spatial model versions are presented in Figure 5.24. Off-site toxicity impacts for all air emissions over the 100-year time period totalled 1.39E-02 DALY for the Urban box model, approximately 6% of the impact due to water emissions evaluated by the Modified Watershed model and 0.01% of water emission impacts evaluated by the Watershed model. Air emissions evaluated by the Non-spatial model totalled 7.09E-03 DALY, approximately 0.01% of water emission toxic impacts as evaluated by the Non-spatial model. Mercury was responsible for 89% of the toxic impacts due to air emissions evaluated by the Urban box and 97% of emissions evaluated by the Non-spatial model.

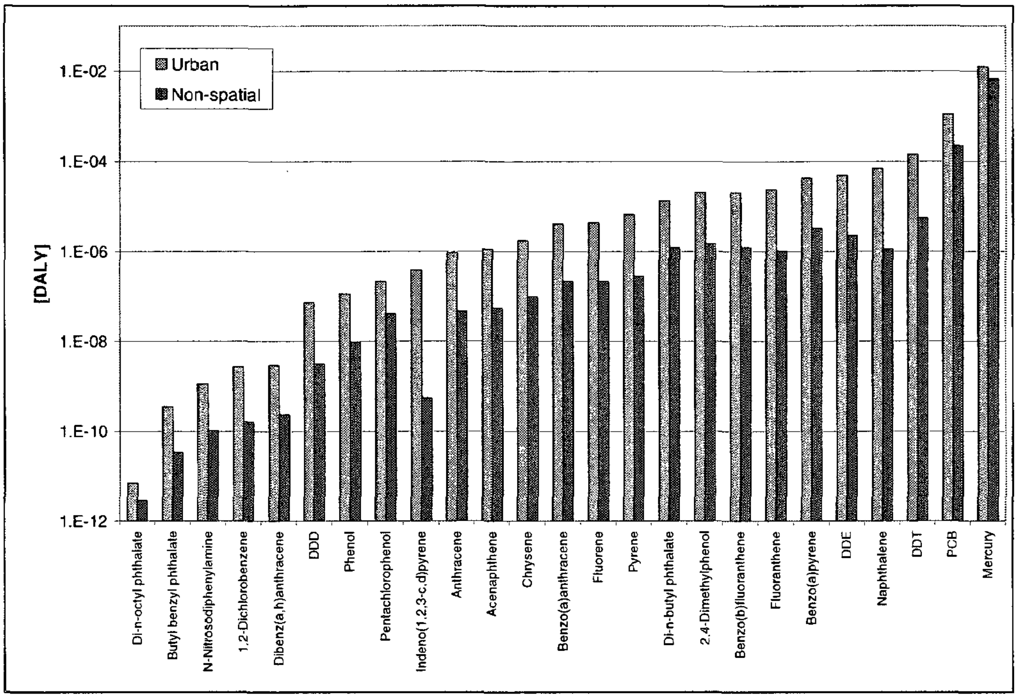


Figure 5.24. Off-site human toxicity impacts associated with MNA air emissions, as evaluated by the Urban box and Non-spatial IMPACT model versions.

5.3.2 Impact assessment of Option 2: Dredging and confined disposal

Figure 5.25 shows the relative contribution to each impact category due to the Remediation Activities. The results indicate that the impacts caused by the Remediation Activities are dominated by the use of the geotextile liner in all impact categories, with the exception of Ionizing Radiation, Ozone Layer Depletion and Terrestrial Ecotoxicity for which sediment transport is the major source of impact. The geotextile liner was modeled as a polypropylene product (CETCO, 2009), which for every kilogram produced, requires 1.02 kilograms of crude oil in addition to numerous other material and energy inputs.

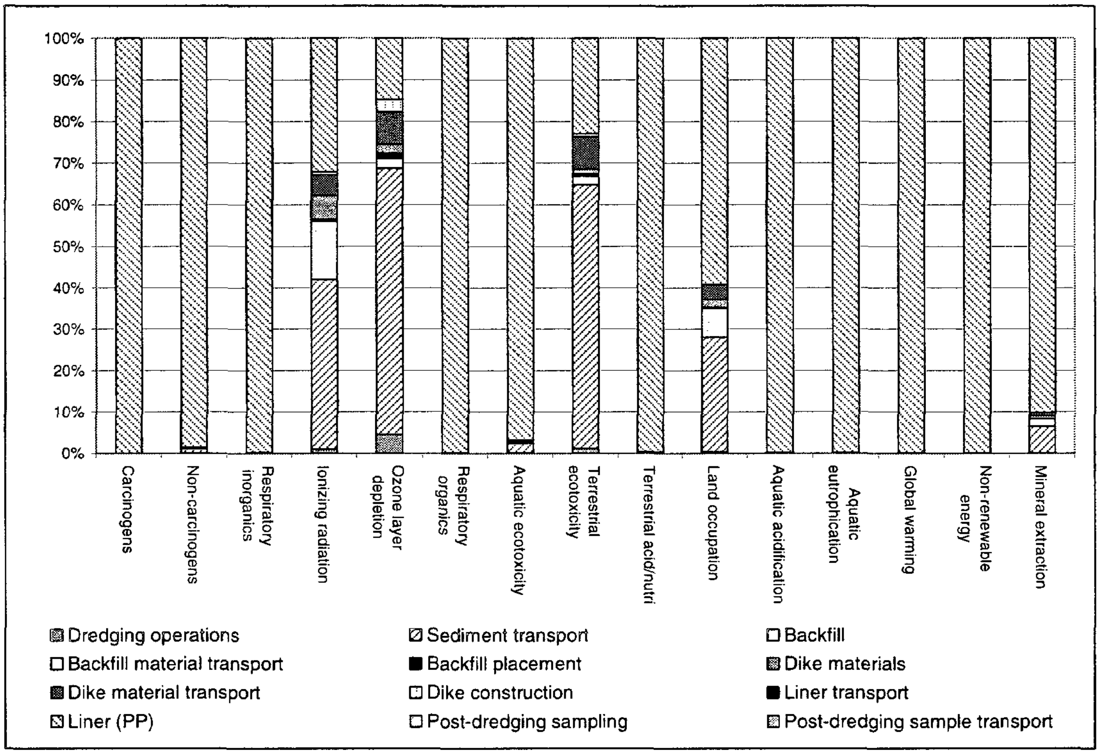


Figure 5.25. Relative contribution of the Remediation Activities processes to IMPACT 2002+ impact categories.

5.3.3 Comparison of impact assessment results considering the MWW site as part of the ecosphere

The midpoint impact category results for MNA, as evaluated by each of the IMPACT models are shown in Figure 5.26, compared with the results for the Remediation Activities. This analysis considers the site to be part of the ecosphere and thus the on-site ecotoxicological impacts due to contaminated sediments are included. Figure 5.27 presents the same results evaluated for the damage impact categories.

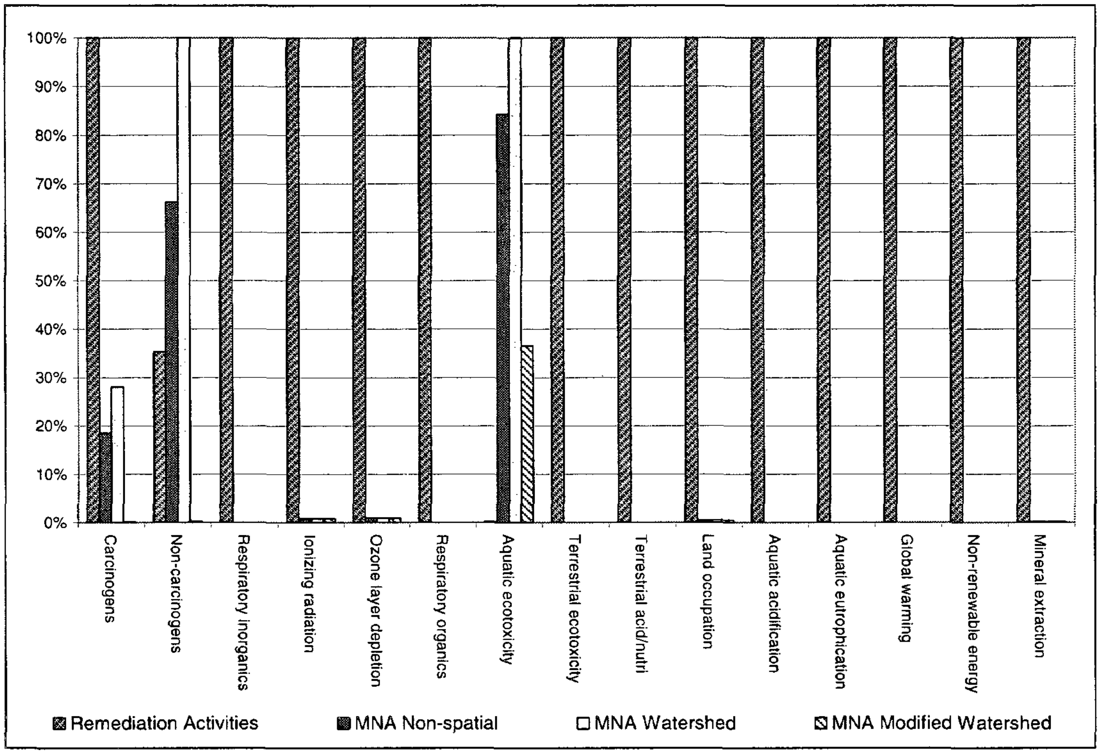


Figure 5.26. Impacts due to the Remediation Activities and MNA as evaluated by the three IMPACT models for IMPACT 2002+ midpoint categories, considering the MWW site to be part of the ecosphere.

Midpoint indicator category results in Figure 5.26 show that the Remediation Activities cause greater impacts than MNA in all categories except for Aquatic Ecotoxicity and Non-carcinogen toxicity. Non-carcinogen toxicity impacts caused by MNA were greater than those caused by the Remediation Activities, except in the case where MNA is evaluated using the Modified Watershed model, in which case this impact is two orders of magnitude smaller (9.28E+05 versus 6.36E+03 kg C_2H_3Cl eq for the Remediation Activities and MNA, respectively).

MNA Aquatic Ecotoxicity impacts were greater than those of the Remediation Activities for all three models. The difference in results for this indicator reflect the different evaluations by the three models of off-site emissions from the contaminated site, with the Watershed model producing the greatest impact for this indicator (1.36E+12 kg TEG eq), followed closely by the Non-spatial model (1.15E+12 kg TEG eq) and then the Modified Watershed model (4.98E+11 kg TEG eq), whose impact is primarily the result of the on-site sediment contamination (4.96E+11 kg TEG eq).

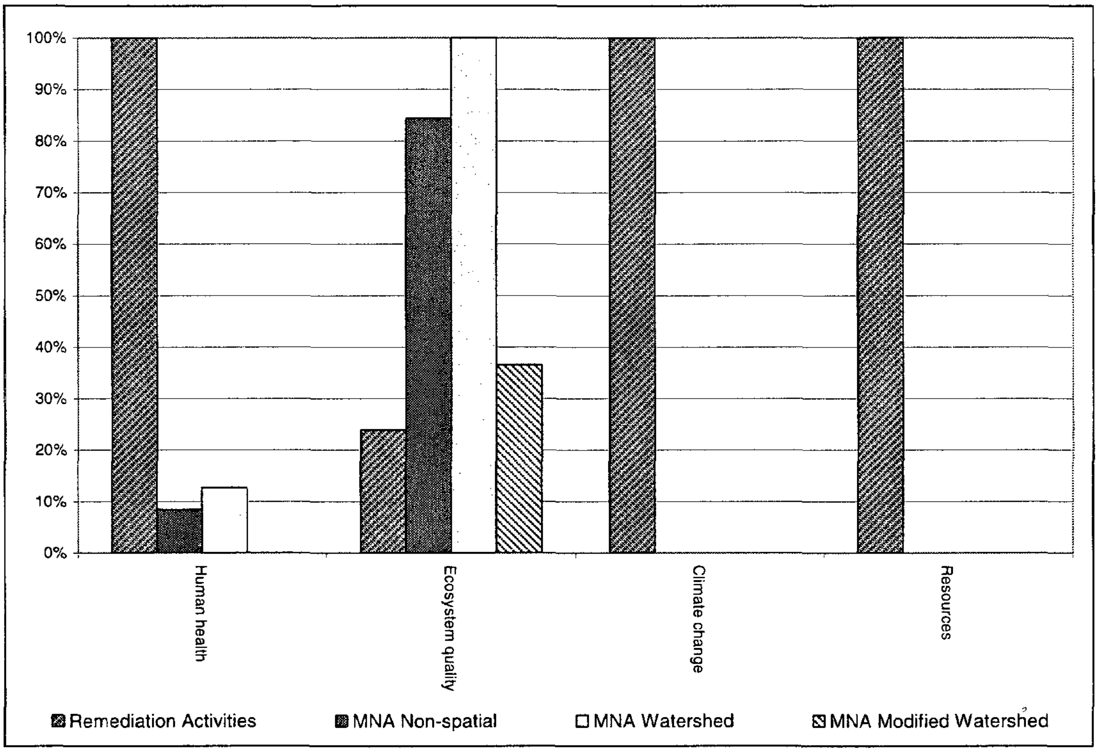


Figure 5.27. Impacts due to the Remediation Activities and MNA as evaluated by the three IMPACT models for IMPACT 2002+ damage categories, considering the MWW site to be part of the ecosphere.

The midpoint indicator results are summarized by the damage indicator results, showing that impacts due to the Remediation Activities are greater than those due to MNA in all categories, except for Ecosystem Quality. The relatively high Terrestrial Acidification/Nutrification midpoint impact (1.52E+07 kg SO_2 eq) due to the fabrication of the polypropylene liner for the Remediation Activities is the main driver for an Ecosystem Quality damage indicator result that is within the same order of magnitude, but still less than that associated with MNA. A small portion (3%) of the Remediation Activities' Ecosystem Quality impacts can be attributed to 5.78E+07 kg TEG eq of Terrestrial Ecotoxicty impacts mostly due to the air emissions of aluminum (36 kg), mercury (1.03 kg) and zinc (2.3 kg) and soil emissions of aluminum (0.9 kg) and zinc (0.13 kg).

It can be noted that although MNA scored higher for Non-carcinogen toxicity with the Non-spatial (1.74E+06 kg C_2H_3Cl eq) and Watershed (2.63E+06 kg C_2H_3Cl eq) models than the Remediation Activities (9.28E+05 kg C_2H_3Cl eq), the higher Carcinogen toxicity score for the Remediation Activities (1.30E+08 kg C_2H_3Cl eq versus 3.64E+07 kg C_2H_3Cl eq for the Watershed model) results in an overall higher Human Health impact for this option.

5.3.4 Comparison of impact assessment results considering the MWW site as part of the technosphere

Midpoint impact category results for MNA, as evaluated by each of the IMPACT models are shown in Figure 5.28, compared with the results for the Remediation Activities. This analysis considers the site to be part of the technosphere and thus the on-site ecotoxicological impacts due to contaminated sediments are not included. Figure 5.29 presents the same results evaluated for the damage impact categories.

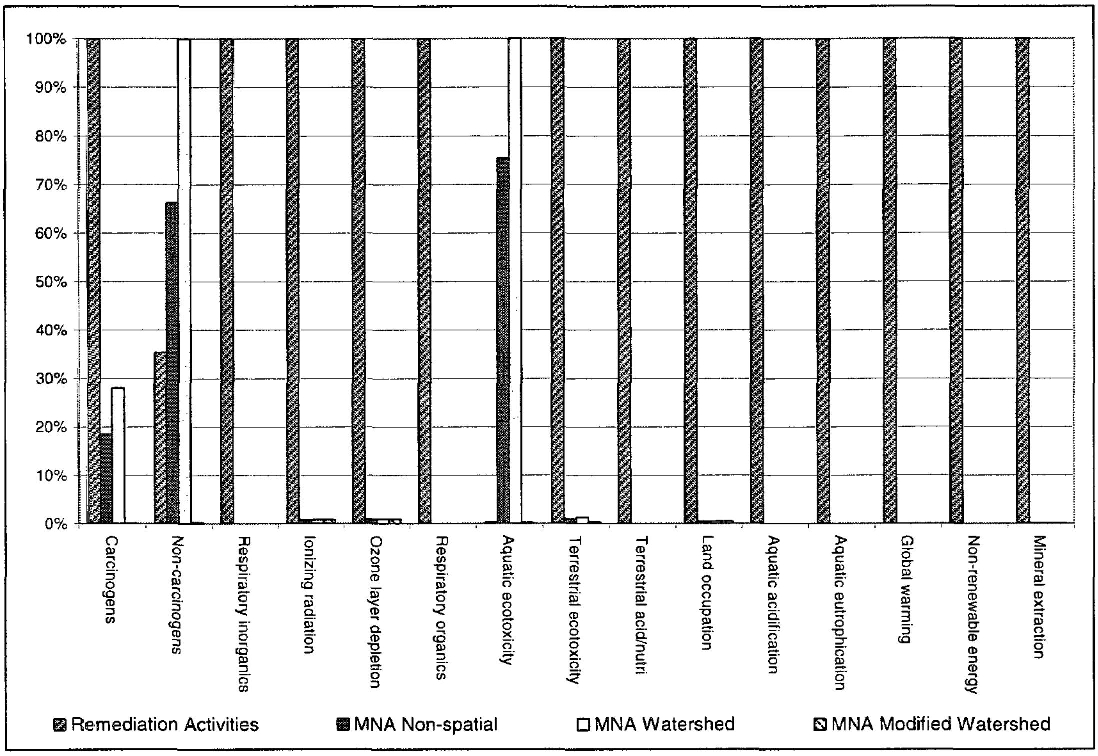


Figure 5.28. Impacts due to the Remediation Activities and MNA as evaluated by the three IMPACT models for IMPACT 2002+ midpoint categories, considering the MWW site to be part of the technosphere.

When considering the site as part of technosphere, midpoint impact category results are identical to those when the site is considered part of the ecosphere for all impact categories except Aquatic Ecotoxicity. The differentiation between model results is more apparent with this impact category when on-site sediment impacts are not considered, as the MNA aquatic ecotoxicological impact evaluated by the Non-spatial model is only 75% that of the Watershed model (6.53E+11 and 8.66E+11 kg TEG eq, respectively) and each of these are two orders of magnitude greater than that of the Modified Watershed model (2.16E+09 kg TEG eq). The Remediation Activities ecotoxicological impact (2.48E+09 kg TEG eq) is approximately equal to the MNA impact evaluated with the Modified Watershed model. It should be noted that for no impact category does MNA have a higher impact than the Remediation Activities, when evaluated using the Modified Watershed model and the site is considered part of the technosphere.

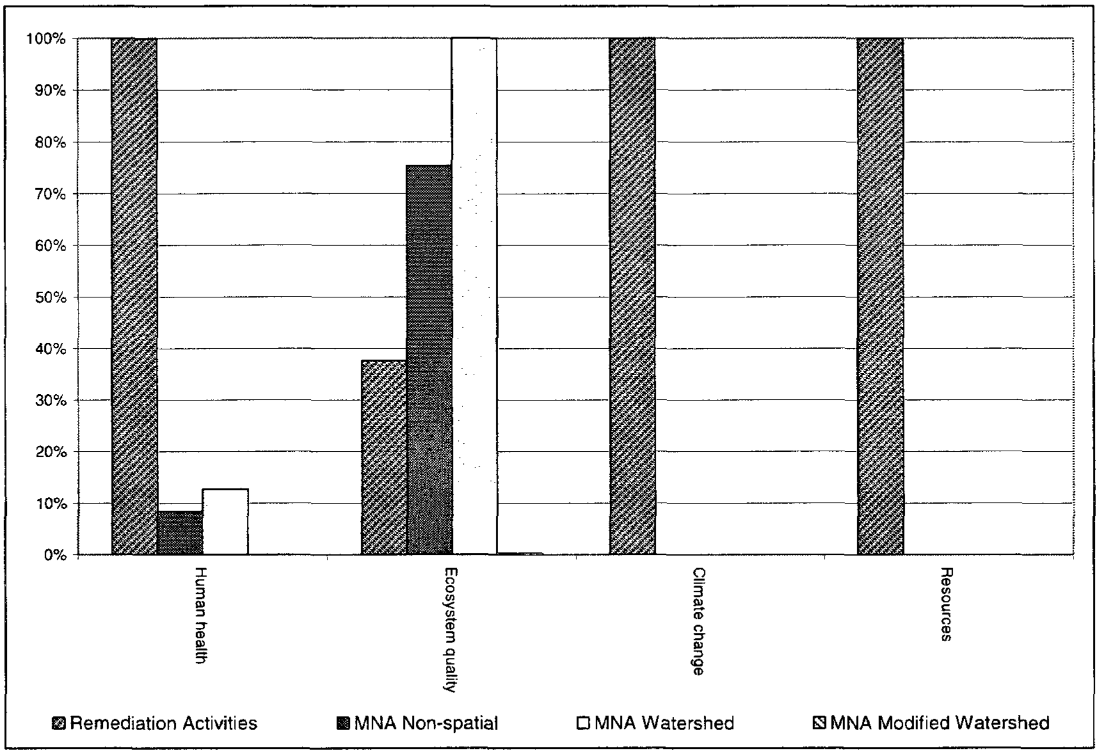


Figure 5.29. Impacts due to the Remediation Activities and MNA as evaluated by the three IMPACT models for IMPACT 2002+ damage categories, considering the MWW site to be part of the technosphere.

When the midpoint indicator impacts are summarized into damage indicator impacts, the results are identical to those for when the site is considered part of the ecosphere, with the Remediation Activities causing greater impacts in all categories except for Ecosystem Quality. The Ecosystem Quality impacts associated with the Remediation Activities amount to 50% of the Non-spatial model MNA impacts and 28% of the Watershed model MNA impacts, but are 148 times greater than those of the Modified Watershed model MNA evaluation. It should be noted that when the site is considered part of the technosphere, MNA, as evaluated by the Modified Watershed model has a smaller impact for all damage indicator categories.

5.3.5 LCIA Summary

Tables 5.10, 5.11, 5.12 and 5.13 indicate the inventory items causing the main impacts for each of the management options and the damage impact category to which they contribute. The quantities for Remediation Activities inventory items in Table 5.11 can be attributed almost entirely to the polypropylene liner.

Table 5.10. Inventory items causing main impacts for MNA according to the Non-spatial IMPACT model.

		Non-spatial	
	Mass	Human Health	Ecosystem Quality
INVENTORY ITEM	[kg]	[DALY]	[PDF-m^2-yr]
On-site metals (t=0)	3.25E+05		2.48E+07
On-site metals (t=100 years)	1.69E+01		
On-site organics (t=0)	1.02E+04		7.94E+04
On-site organics (t=100 years)	2.33E-06		
Off-site metal emissions to water	2.90E+05	6.76E+01	3.28E+07
Off-site metal emissions to air	8.71E+00	6.85E-03	5.12E+02
Off-site organics emissions to water	3.52E+03	5.45E-03	2.66E+03
Off-site organics emissions to air	4.80E+02	2.43E-04	7.78E+00

Table 5.11. Inventory items causing main impacts for MNA according to the Watershed IMPACT model

		Watershed	
	Mass	Human Health	Ecosystem Quality
INVENTORY ITEM	[kg]	[DALY]	[PDF-m^2-yr]
On-site metals (t=0)	3.25E+05		2.48E+07
On-site metals (t=100 years)	1.69E+01		
On-site organics (t=0)	1.02E+04		7.94E+04
On-site organics (t=100 years)	2.33E-06		
Off-site metal emissions to water	2.90E+05	1.02E+02	4.35E+07
Off-site metal emissions to air	8.71E+00	1.24E-02	5.32E+03
Off-site organics emissions to water	3.52E+03	7.42E-03	2.80E+03
Off-site organics emissions to air	4.80E+02	1.51E-03	4.45E+01

Table 5.12. Inventory items causing main impacts for MNA according to the Modified Watershed IMPACT model

		Modified Watershed	
	Mass	Human Health	Ecosystem Quality
INVENTORY ITEM	[kg]	[DALY]	[PDF-m^2-yr]
On-site metals (t=0)	3.25E+05		2.48E+07
On-site metals (t=100 years)	1.69E+01		
On-site organics (t=0)	1.02E+04		7.94E+04
On-site organics (t=100 years)	2.33E-06		
Off-site metal emissions to water	2.90E+05	2.29E-01	1.03E+05
Off-site metal emissions to air	8.71E+00	1.24E-02	5.32E+03
Off-site organics emissions to water	3.52E+03	9.02E-05	6.82E+01
Off-site organics emissions to air	4.80E+02	1.51E-03	4.45E+01

Table 5.13. Inventory impacts causing main impacts for the Remediation Activities.

			Human health	Ecosystem quality	Climate change	Resources
	Amount	Unit	[DALY]	[PDF-m^2-yr]	[kg CO_2 eq]	[MJ primary]
Aromatic hydrocarbons	3.67E+04	[kg to air]	3.64E+02	5.30E+03		
Nitrogen oxides	2.26E+06	[kg to air]	2.01E+02	1.29E+07		
Sulfur dioxide	2.75E+06	[kg to air]	1.50E+02	2.86E+06		
Zinc	5.27E+00	[kg to soil]	1.11E-02	2.47E+05		
Aluminum	6.27E+01	[kg to air]	8.53E-05	6.43E+04		
Carbon dioxide	1.02E+09	[kg to air]			1.02E+09	
Energy, various sources	3.51E+10	[MJ]				3.51E+10

5.4 Interpretation

5.4.1 Sensitivity/Uncertainty

The results of the LCA must be interpreted in the context of the sensitivity and uncertainty associated with the models used to produce them. This section presents the results of the MNA model sensitivity analysis as well as a discussion regarding the sensitivity and uncertainty associated with the IMPACT models.

5.4.1.1 MNA model sensitivity analysis

The results of the MNA model sensitivity analysis for the six substances tested are presented in Table 5.14. The model output for water and air emissions was most sensitive to changes in the total suspended sediment concentration, the organic carbon content of the sediments and suspended sediments, and the organic carbon partitioning coefficient. These parameters showed approximately identical sensitivity, as they are each implicated in the partitioning of contaminants between the suspended sediment and water column and determine how much of the contaminant is available for volatilization and how much will be transported off-site by water flow adsorbed to suspended sediment. A 10% change in these parameters caused a greater than 10% change in the mass of water emissions for all six substances tested except DDT, which experienced a 6.3% change. A 10% change in these parameters caused a greater than 10% change in the mass of air emissions for all six substances tested except copper, which is not volatile. Mercury emissions were notably more sensitive to these parameters, exhibiting a 35% increase in emissions to air and 14% increase in water emissions with a 10% decrease in parameter values. This sensitivity is likely due to the fact that it has both a high sediment partitioning coefficient (7.15E+03 at 3.5% organic carbon) and is susceptible to volatilization (Henry law constant of 8.61E+02 Pa-m^3/mol), such that any change affecting the partitioning of mercury between water and suspended sediment results in a change in both water and air emissions. Substances with lower Henry's law constants will remain dissolved in the water column if they are not adsorbed to suspended sediment and water emissions from the MWW site will not be greatly affected.

Table 5.14. MNA model sensitivity analysis results showing the percent change in model result due to a 10% change in parameter value.

Parameter	**1,2-Dichlorobenzene**	**DDT**	**Indeno(1,2,3-cd)pyrene**	**Phenol**	**Copper**	**Mercury**
Wind speed						
Water emissions	0.00%	0.65%	0.00%	0.02%	0.00%	11.69%
Air emissions	0.00%	6.61%	9.99%	13.93%	0.00%	19.01%
Time-integrated sediment concentration	0.00%	0.02%	0.00%	0.02%	0.00%	0.02%
Organic Carbon						
Water emissions	11.63%	6.30%	10.14%	11.12%	10.63%	14.12%
Air emissions	10.66%	10.35%	10.14%	11.14%	0.00%	35.29%
Time-integrated sediment concentration	0.76%	0.00%	0.00%	8.68%	1.13%	0.00%
Total suspended solids						
Water emissions	11.64%	6.30%	10.14%	11.12%	10.62%	14.12%
Air emissions	10.67%	10.35%	10.14%	11.14%	0.00%	35.29%
Time-integrated sediment concentration	0.78%	0.00%	0.00%	8.70%	0.96%	0.00%
Porosity						
Water emissions	0.00%	0.01%	0.00%	0.01%	0.03%	0.01%
Air emissions	0.00%	0.01%	0.00%	0.01%	0.00%	0.01%
Time-integrated sediment concentration	0.64%	0.00%	0.00%	7.79%	0.38%	0.00%
K_{OC}						
Water emissions	11.63%	6.30%	10.14%	11.12%	10.63%	14.12%
Air emissions	10.66%	10.35%	10.14%	11.14%	0.00%	35.29%
Time-integrated sediment concentration	0.76%	0.00%	0.00%	8.68%	0.80%	0.00%

Table 5.14. MNA model sensitivity analysis results (continued).

Parameter	1,2-Dichlorobenzene	DDT	Indeno(1,2,3-cd)pyrene	Phenol	Copper	Mercury
Henry's law constant						
Water emissions	0.00%	0.33%	0.00%	0.02%	0.00%	0.24%
Air emissions	0.00%	3.37%	9.98%	13.89%	0.00%	0.38%
Time-integrated sediment concentration	0.00%	0.00%	0.00%	0.02%	0.00%	0.01%
Water decay						
Water emissions	0.04%	2.33%	9.79%	2.58%	0.00%	0.00%
Air emissions	0.04%	2.33%	9.79%	2.58%	0.00%	0.00%
Time-integrated sediment concentration	0.00%	0.00%	0.01%	2.02%	0.00%	0.00%
Sediment decay						
Water emissions	0.01%	1.44%	3.27%	0.00%	0.00%	0.00%
Air emissions	0.01%	1.44%	3.27%	0.00%	0.00%	0.00%
Time-integrated sediment concentration	7.40%	2.34%	3.35%	10.99%	0.00%	0.00%
Pollution return flow						
Water emissions	5.77%	3.40%	9.05%	2.75%	0.03%	4.05%
Air emissions	4.70%	7.33%	1.05%	8.06%	0.00%	6.61%
Time-integrated sediment concentration	0.32%	0.01%	0.00%	6.31%	0.97%	0.02%

For these same reasons mentioned above, mercury air and water emissions showed high sensitivity to the wind speed used in the MNA model. Phenol, indeno(1,2,3-cd)pyrene and DDT also showed sensitivity to a 10% decrease in wind speed with 14%, 10% and 6.6% decreases in air emissions, respectively. Air emissions of these substances showed similar sensitivity to changes in the Henry's law constant. Mercury and 1,2-dichlorobenzene were affected little by this change, as their Henry's law constant values are already relatively high.

The model outputs showed little sensitivity to sediment porosity, with the exception of the time-integrated concentration of phenol. Sediment porosity determines the amount of water available for partitioning in the sediments. Phenol, with the lowest sediment partitioning coefficient is the most sensitive to changes in this parameter value.

Indeno(1,2,3-cd)pyrene showed a 9.8% increase in water and air emissions with a 10% decrease in water decay rate. Phenol and DDT showed lower sensitivity to this parameter, with a 2.58% and 2.33% increase in emissions. The time-integrated sediment concentrations of all organic substances tested were sensitive to the sediment decay rate, with phenol showing high sensitivity (11% change), followed by 1,2-dichlorobenzene (7.4%), indeno(1,2,3-cd)pyrene (3.35%) and DDT (2.34%).

The water and air emissions were sensitive to the return flow factor for all substances tested except copper. The return flow factor directly affects the water residence time in MWW, thereby influencing the relative quantity of contaminant available for volatilization or emission via water. With a 10% decrease in return flow, the flow rate was effectively increased, lowering the water residence time, thereby lowering volatilization from MWW and increasing water emissions for most substances. Copper, on the other hand, is not volatile and its water concentration profile indicated that it was not long-lived in the water column (see Table 5.3: a ten-fold concentration decrease in 7 days). As such, a change in flow had little effect on the total mass of water emissions over the 100-year time period.

With the exception of a few metals, no significant difference can be noted in the evaluation of mass emitted or time-integrated contaminant concentration between the 50- and 100-year time frames (see Table 5.7). A time frame of 100 years resulted in a 16% increase in time-integrated sediment concentration for antimony, relative to the 50-year value and a 7% increase in the mass of cadmium and lead emitted via water.

5.4.1.2 MNA and IMPACT model uncertainty

No formal uncertainty analysis of the MNA or IMPACT models was carried out, however a descriptive, order-of-magnitude consideration of uncertainty is presented here to assist in interpretation of the results.

The MNA model makes simplifying assumptions regarding the mass transfer relationships between the sediment layers and the water column. Uniform compartments with concentrations governed by constant rates is a simplification that may stray significantly from reality such that small-scale variations may be important for contaminant transport. Moreover, processes or factors that have not been considered, such as storm event mixing, vertical water flow due to sediment compaction and the influence of external contaminant loading might play a large role in determining emissions from the MWW site.

For the geographic scale at which multimedia models such as IMPACT evaluate contaminant fate, the requirement for simplifying assumptions increases, but the uncertainty associated with the model can still be significant (McKone *et al.*, 2001; Huijbregts *et al.*, 2003; Webster *et al.*, 2004).

Variation in parameter values (parameter uncertainty) is a significant source of uncertainty for both models. In the previous section, the MNA model was shown to be sensitive to parameters that exhibit natural variation across several orders of magnitude. In particular, partitioning coefficients and decay rates are highly dependent on local conditions and are frequently cited as great sources of uncertainty for multimedia fate and exposure models (McKone, 1993; Eisenberg & McKone, 1998; Huijbregts *et al.*, 2000; Webster *et al.*, 2004; Rosenbaum, 2006). Moreover, the lack of consideration for metal speciation introduces further uncertainty into the model results.

Given the potential for considerable uncertainty in the model output, it is proposed for the present study that differences in LCA results of less than two orders of magnitude be considered equal. This rule-of-thumb will not be applied to the differences noted in characterization factors however. The IMPACT models use the same mass balance relationships and chemical data, such that differences in model output can be attributed to the introduced differences in the spatial differentiation of landscape parameters.

5.4.2 LCA Interpretation

The Remediation Activities were evaluated as causing greater impacts than MNA, whether the MWW site is considered part of the ecosphere or technosphere. As indicated in Section 5.4.1.2, to account for model uncertainty, all results that differed by less than two orders of magnitude were considered to be equal.

For the ecosphere perspective, the Remediation Activities cause greater impacts in the Climate Change and Resource damage impact categories and there is no significant difference in Human Health or Ecosystem Quality impacts between the options, except where MNA is evaluated using the Modified Watershed model, in which case the MNA Human Health impacts are less than those of the Remediation Activities. Taking the technosphere perspective, the result is the same, except both the Human Health and Ecosystem Quality impacts of the Modified Watershed MNA are less than those of the Remediation Activities.

When considering the MWW site as part of the ecosphere, the MNA impacts were dominated by the off-site emissions for the Non-spatial and Watershed model evaluations, but the on-site contamination was the most important impact for the Modified Watershed model. Metals accounted for a large majority of MNA toxicity and ecotoxicity impacts for all models, both on-site and off-site, with zinc implicated as the main contaminant of concern, followed by copper. Mercury accounted for most of the impacts due to off-site emissions to air, however these impacts were not significant contributors to either human toxicity or ecotoxicity.

The polypropylene liner was responsible for most of the impacts caused by the Remediation Activities, followed by sediment transport.

Little comparison is needed between MNA and the Remediation Activities with respect to Climate Change and Resources, as MNA had relatively little contribution in these damage impact categories. However, within the Human Health and Ecosystem Quality impact categories, emissions due to fabrication of the polypropylene liner were significant enough to equal both the off-site and on-site MNA impacts for the Non-spatial and Watershed models. The liner emissions caused greater Human Health and Ecosystem Quality impacts than the Modified Watershed MNA assessment. Zinc emissions to soil and air emissions of aromatic hydrocarbons, nitrogen oxides, sulfur dioxide, and aluminum make up most of the liner emissions' impact.

6 Complementary Results

This section describes the influence of the various processes considered by the MNA model in determining contaminant emissions and sediment concentrations at the MWW site.

6.1 LCI Interpretation: MNA model contribution analysis

The total mass transferred (or transformed) by each mass balance process can be calculated by replacing the relevant concentration term in the mass balance equation with the appropriate integral of the concentration profile. For example, the calculation to determine the total mass subjected to decay over 100 years in the water column is shown below in Equation 6.1:

$$\text{mass decay} = k_w \cdot V_w \cdot \int_0^{100} c_w(t)\,dt \qquad \text{(Eq. 6.1)}$$

where:

mass decay = mass subjected to decay over 100 years in the water column [kg]
k_w = decay rate in water [1/day]
V_w = average volume of water [m^3]
$c_w(t)$ = concentration in water as a function of time [kg/m^3]

Using the quantities calculated as above, removal processes were evaluated according to the total mass removed by each process divided by the sum of all removal processes for the water column and mixed sediment layer. Tables 6.1 and 6.2 present the relative contribution of each removal process within the water column and mixed sediment layer, respectively.

Table 6.1. Water column removal process contribution analysis showing relative mass removed by decay, volatilization and outflow. Diffusion and settling were not significant removal processes for any contaminant (contributing to greater than 0.1%).

Substance	**Decay**	**Volatilization**	**Outflow**
1,2-Dichlorobenzene	0.3%	52.5%	47.2%
2,4-Dimethylphenol	17.3%	43.6%	39.2%
Acenaphthene	5.5%	49.8%	44.8%
Anthracene	34.5%	34.5%	31.0%
Benzo(a)anthracene	52.6%	0.0%	47.4%
Benzo(a)pyrene	24.8%	0.0%	75.2%
Benzo(b)fluoranthene	4.8%	0.0%	95.2%
Butylbenzylphthalate	16.1%	0.0%	83.8%
Chrysene	52.6%	0.0%	47.4%
DDD	0.0%	0.0%	100.0%
DDE	9.8%	47.5%	42.7%
DDT	0.4%	0.2%	99.4%
Dibenzo(a,h)anthracene	4.5%	0.0%	95.5%
Di-n-butylphthalate	4.6%	50.2%	45.2%
Di-n-octylphthalate	4.2%	0.0%	95.8%
Fluoranthene	30.6%	0.3%	69.1%
Fluorene	0.8%	52.2%	47.0%
Indeno(1,2,3-cd)pyrene	0.4%	0.0%	99.6%
Naphthalene	3.4%	50.8%	45.7%
N-Nitrosodiphenylamine	2.3%	51.4%	46.3%
PCB (total)	0.3%	0.0%	99.7%
Pentachlorophenol	52.6%	0.0%	47.4%
Phenol	52.6%	0.1%	47.3%
Pyrene	52.5%	0.2%	47.3%
Antimony	0.0%	0.0%	100.0%
Arsenic	0.0%	0.0%	100.0%
Cadmium	0.0%	0.0%	100.0%
Copper	0.0%	0.0%	100.0%
Lead	0.0%	0.0%	100.0%
Mercury	0.0%	1.2%	98.8%
Nickel	0.0%	0.0%	100.0%
Zinc	0.0%	0.0%	100.0%

Table 6.2. Mixed sediment layer removal process contribution analysis showing relative mass removed by decay, resuspension and burial. Diffusion was not a significant removal process for any contaminant (contributing to greater than 0.1%).

Substance	**Decay**	**Resuspension**	**Burial**
1,2-Dichlorobenzene	69%	12%	19%
2,4-Dimethylphenol	97%	1%	2%
Acenaphthene	99%	0%	0%
Anthracene	47%	21%	32%
Benzo(a)anthracene	37%	25%	38%
Benzo(a)pyrene	42%	23%	35%
Benzo(b)fluoranthene	38%	25%	37%
Butylbenzylphthalate	90%	4%	6%
Chrysene	28%	29%	43%
DDD	23%	31%	46%
DDE	23%	31%	46%
DDT	23%	31%	46%
Dibenzo(a,h)anthracene	30%	28%	42%
Di-n-butylphthalate	99%	1%	1%
Di-n-octylphthalate	82%	7%	11%
Fluoranthene	47%	21%	32%
Fluorene	86%	6%	8%
Indeno(1,2,3-cd)pyrene	32%	27%	41%
Naphthalene	87%	5%	8%
N-Nitrosodiphenylamine	79%	9%	13%
PCB (total)	6%	38%	56%
Pentachlorophenol	79%	9%	13%
Phenol	97%	1%	2%
Pyrene	18%	33%	49%
Antimony	0%	40%	60%
Arsenic	0%	40%	60%
Cadmium	0%	40%	60%
Copper	0%	40%	60%
Lead	0%	40%	60%
Mercury	0%	40%	60%
Nickel	0%	40%	60%
Zinc	0%	40%	60%

Metals do not decay, and with the exception of mercury, do not volatilize, such that all metals removal from the water column could be attributed to outflow. A small percentage of mercury was predicted to leave the MWW site as an air emission.

Outflow was a major removal process for most other substances, with the lowest removal contribution from advection equalling 39.2% (2,4-dimethylphenol) and 12 of the remaining 30 organic substances showing outflow responsible for at least 50% of mass removed from the water column.

For the remainder of mass removal from the water column, most organic contaminants could be characterized as either being degraded or volatilized, with only four substances (2,4-dimethylphenol, anthracene, benzoic acid and DDE) experiencing significant (10% or greater) removal from both processes. Diffusion did not account for greater than 0.1% of removal for any substance.

For the mixed sediments, decay was generally an important removal process for the organic contaminants, contributing to at least 23% of removal in all cases except PCB and pyrene, and up to nearly 100% for four substances (2,4-dimethylphenol, acenaphthene, di-n-butylphthalate and phenol). The remainder of contaminant removal from the mixed sediments after decay was consistently split 40% - 60% between resuspension and burial, respectively, reflective of their rates.

6.2 Contaminant fate

The concentration profile expressions were also able to provide the final mass in the deep sediment compartment. Using this quantity and the total mass transferred offsite by various processes, the fate of contaminants in MWW as predicted by the MNA model can be evaluated. Table 6.3 shows the relative mass of contaminant removed from the MWW site after 100 years via water emission, air emission or decay, as well as the relative mass remaining in the deep sediment. After 100 years, no contaminant mass remained in the mixed sediments. If substances were not degraded or transferred to the water column, then burial moved them to the deep sediments.

Table 6.3. Fate of MWW site contaminants: the percentage of the original contaminant mass remaining in the deep sediments or removed via water emission, air emission or decay.

Substance	Water emission	Air emission	Decay	Deep sediments
1,2-Dichlorobenzene	45%	54%	1%	0%
2,4-Dimethylphenol	39%	44%	17%	0%
Acenaphthene	19%	48%	33%	0%
Anthracene	3%	23%	74%	0%
Benzo(a)anthracene	0%	0%	100%	0%
Benzo(a)pyrene	0%	0%	100%	0%
Benzo(b)fluoranthene	1%	0%	99%	0%
Butylbenzylphthalate	17%	0%	83%	0%
Chrysene	0%	0%	100%	0%
DDD	2%	0%	98%	0%
DDE	1%	24%	75%	0%
DDT	9%	1%	90%	0%
Dibenzo(a,h)anthracene	0%	0%	100%	0%
Di-n-butylphthalate	36%	56%	8%	0%
Di-n-octylphthalate	0%	0%	100%	0%
Fluoranthene	5%	0%	95%	0%
Fluorene	11%	43%	46%	0%
Indeno(1,2,3-cd)pyrene	1%	0%	99%	0%
Naphthalene	38%	54%	8%	0%
N-Nitrosodiphenylamine	39%	56%	4%	0%
PCB (total)	10%	1%	77%	12%
Pentachlorophenol	42%	0%	58%	0%
Phenol	47%	0%	53%	0%
Pyrene	2%	0%	98%	0%
Antimony	97%	0%	0%	3%
Arsenic	47%	0%	0%	53%
Cadmium	14%	0%	0%	86%
Copper	82%	0%	0%	18%
Lead	17%	0%	0%	83%
Mercury	18%	11%	0%	72%
Nickel	97%	0%	0%	3%
Zinc	96%	0%	0%	4%

With the exception of PCB, no organic contaminants were predicted to remain in the deep sediments (and hence MWW site) after 100 years.

For the organic substances at MWW site, decay is the best first descriptor of fate, ranging from 1 to 100%. Where decay does not dominate, neither the emissions via air or emissions via water are likely to be significantly greater than the other, with the exception of the alcohols pentachlorophenol, phenol and benzyl alcohol, this latter being the only substance for which an emission is responsible for more than 60% of an organic contaminant mass' fate with respect to MWW.

The fate of metals in MWW can be dominated by either emission to Puget Sound via water or burial to the deep sediments. Most metals show some mix of these fate pathways, except antimony, nickel and zinc, which are almost entirely removed from MWW via water emission.

7 Discussion

7.1 MNA model

The primary function of the MNA model for the present study was to generate the LCI for the MNA management option. It is concerning that metals represented a major component of the on-site contamination and off-site emissions, given that such models are best suited to evaluating the mass transfer of organic substances, using partitioning coefficients as a key parameter. The ability of the MNA model to accurately describe the fate of metals is dependent on the ability of the empirically-derived metal partitioning coefficients from CalTOX and USEtox to describe the behaviour of various metal species. The validity of these partitioning coefficients and their applicability to the MWW site was not investigated and adds a significant amount of uncertainty to the model results.

With more data available, it might be possible to account for metal speciation at the MWW site using an equilibrium speciation model such as MINTEQA2 (USEPA, 2006). Although designed for use in aqueous systems, it may be possible to extend the use of the MINTEQA2 model to the porewater of the sediments. Of particular importance with respect to data requirements would be the need to know the quantity of acid volatile sulfides (AVS) available, which have the capacity to form non-bioavailable sulfides with copper, lead, cadmium, zinc, nickel, mercury and silver in sediments. Numerous studies indicate that when the molar concentration of AVS is in excess of the molar sum of the concentrations of cationic metals, these metals do not cause toxic effects (Di Toro *et al.*, 1992; Ankley *et al.*, 1996; Chapman *et al.*, 1998; Simpson, 2001). It should be noted however that AVS data were not available in the MWW sampling reports (Anchor & Foster Wheeler, 2001).

A secondary function of the MNA model was demonstrated in assessing exceedances of the Washington State surface water quality criteria and CBNT sediment quality objectives. It may be sufficient to use average sediment concentration values in the model in order to assess surface water quality, however for the purposes of assessing remediation progress, a more precise spatial description of sediment quality would be required and the individual sections of MWW would need to be modeled separately. This task would require creating a mass balance for the mixed and deep sediments for each section, with one mass balance for the water column that considers transfers with the mixed sediments of all the sections. The resulting system would have two equations for each section, representing the sediment layers plus one additional (long) equation for the water column. Computational requirements would certainly be increased, but given that MAPLE generally took no longer than 30 seconds to solve each system of three equations used in the present study (using a computer with a 1.8 GHz processor), the task is likely well within the capacities of a regular desktop computer.

The MNA model is a simplified mass balance model and does not consider certain processes that could be important in determining concentrations in the sediment and overlying water column. Of note, vertical water flow due to sediment compaction has not been considered, nor has sediment and water mixing due to wind and storm events. Given the uncertainties associated with the model, estimates of compliance with water quality criteria or sediment quality objectives should be interpreted with caution.

An additional function that the MNA model could serve would be the performance evaluation of enhanced natural recovery as a management option, in which sediments with minor chemical exceedances are covered with a thin cap to improve habitat conditions and limit the transfer of contaminants to the water column. For such an evaluation, a similar MNA model to the one in the present study could be used, considering the uncontaminated surface layer as the mixed sediments and modeling the lightly contaminated sediments under the thin cap as the deep sediments layer.

7.2 Sediment ecotoxicity factors

The validity of the sediment ecotoxicity effect factors developed using the Equilibrium Partitioning method was not tested in the present study, however some results raised questions as to their appropriateness for assessing sediment contamination.

The use of equilibrium partitioning to calculate sediment ecotoxicity factors resulted in some noteworthy inversions with respect to aquatic ecotoxicity in the case of several hydrophobic compounds with high aquatic toxicity. The corresponding low sediment ecotoxicity did not correlate with the SQO set for the CBNT Superfund site, for which one might expect similar rankings of substance toxicity. Agreement with SQO is not necessary though for LCIA indicators, given that environmental quality criteria derived from risk assessment data typically seek to protect the most sensitive species whereas LCIA indicators look to describe the average species response to a toxicant.

Evidence in the literature indicates that in some cases, strong carbon binding can mitigate sediment toxicity, particularly for PAH (Paine *et al*, 1996; Naes *et al.*, 1999; Lohmann *et al.*, 2004; Neff *et al.*, 2005), supporting the use of equilibrium partitioning to determine sediment ecotoxic effects from corresponding aquatic data. However, evidence from Lu *et al.* (2004) suggests that as hydrophobicity increases, sediment ingestion becomes a more important PAH exposure route for certain benthic species, contradicting the porewater exposure route assumption used by the Equilibrium partitioning method. The use of the EqP method for determining soil quality standards from aquatic quality standards was found to be problematic by van Beelen *et al.* (2003), in that the method risked under- or over-estimating the appropriate 5% Hazardous Concentration (HC5) values by up to a factor of 20 in some cases. Despite these problems, van Beelen *et al.* (2003) advocated the use the EqP method for determining soil quality standards from aquatic quality standards, when only limited terrestrial toxicity data are available.

For the MWW site, the LCIA results indicated that metals were responsible for the bulk of the impact to the sediments. This result is problematic as the EqP method was not originally intended for metals (Di Toro *et al.*, 1991). Partitioning of the metals between the sediment and porewater is described in the present study using the empirically-derived partitioning coefficients available in CalTOX (McKone *et al.*, 2001) and USEtox (Rosenbaum *et al.*, 2008), but no consideration has been made for speciation. It is unclear to what extent this shortcoming affects the ability to compare this toxicity indicator to the IMPACT toxicity indicators, which also have not considered speciation in their development. For the moment, it is assumed that the use of common parameter values for metals within each model will at least allow for reasonable comparisons between processes that produce metal emissions.

As appropriate toxicity data for benthic species are lacking (Huijbregts, 2007), it is recommended that the sediment ecotoxicity effect factors developed here be considered as "interim" effect factors, to be used with caution. Should risk assessment results suggest that highly hydrophobic contaminants are bioavailable to benthic organisms, then the effect factors or their interpretation should be adjusted appropriately.

7.3 Spatial differentiation

7.3.1 Water emissions

Not surprisingly, water residence time was found to be a main driver in determining the different ecotoxicity impacts evaluated by water emissions between IMPACT models and along with substance properties such as ecotoxicity and persistence, could explain differences between contaminants. This result is related to the fact that the watershed tested flowed into the ocean, where no further ecotoxic impact occurred, such that the emission zone parameters were the primary determinants of impacts.

The result might be different for emissions occurring in watersheds further from the ocean, with other watersheds downstream. In such cases, substances would experience a longer total effective water residence time, and it would be expected that less of a difference would be apparent between the Watershed model and a Modified Watershed model, particularly for substances with short half-lives in water that quickly complete their exposure pathway. For substances that are persistent in water, differences between the models, including the Non-spatial model, would then be related to the difference in residence times.

Human toxicity impacts were similarly affected by water residence time, and differences were compounded by the fact that the Modified Watershed emission zone is not used as a drinking water source. An inland emission, occurring in a water body considered by all model versions to be a drinking water source would exhibit less difference between the models.

These results suggest that emissions to freshwater in coastal regions or emissions to freshwater of persistent substances will show the most benefit of spatial differentiation efforts. The effect of spatial differentiation in reducing the characterization factors for metal emissions witnessed in the present study is of particular interest. Impacts due to metals can frequently dominate toxicity and ecotoxicity results (Gloria *et al*, 2006; Haye *et al.*, 2007) and spatial differentiation that more accurately reflects the fate of metals in the environment will have serious implications for LCA results in which these emissions dominate.

7.3.2 Air emissions

The differences in mass transfer of contaminants from the air to the soil and water explained a great deal of the differences between the Urban box and Non-spatial IMPACT models in assessing the ecotoxicity impacts due to air emissions. Precipitation and runoff rates as well as the relative sizes of water and soil compartments were deciding parameters. Much of the difference between the human toxicity impact assessment by the two models is due to the fact that the contaminants found at the MWW site have ingestion rather than inhalation as a primary exposure route. Consequently, considering the population density of the city of Tacoma with the Urban box version of IMPACT did little in deciding differences in contaminant intake fractions, however the agricultural output of the regions surrounding the emission zone appeared to be significant, with exposed produce playing a defining role.

7.4 Site-specificity affects the evaluation of impacts

The different model results caused by spatial differentiation are apparent in the comparison of remediation options. The use of site-specific data to produce the Modified Watershed version resulted in a ranking difference between the options for the Human Health impact category when taking the ecosphere perspective and for both the Human Health and Ecosystem Quality impact categories when taking the technosphere perspective. While the Remediation Activities had approximately equal impact to MNA in these categories for the Non-spatial and Watershed versions, MNA had less impact when evaluated by the Modified Watershed version. Depending on the weighting of these impact categories and the weighting of the LCA results amongst other criteria (e.g., cost, risk assessment results) in the decision-making process, the introduction of site-specific data into the LCIA method could conceivably change the decision taken for a contaminated site remediation project.

Spatial differentiation of the contaminated site emissions raises the question as to whether there are processes associated with the Remediation Activities for which the emission impacts might be mitigated (or exacerbated) if they were evaluated using a spatially differentiated model.

The dominant impacts for the Remediation Activities come from the consumption of crude oil to produce the CDF liner and large quantities of diesel required for construction and transport processes. Given the global nature of oil production and distribution, it is assumed here that the associated processes are well described in a generalized fashion and that spatial differentiation would not reveal great differences in the impacts caused by their emissions.

However, Humbert *et al.* (2009) demonstrated with a LCA of diesel fuel that a spatially differentiated model might find differences up to six orders of magnitude in the intake fraction associated with emissions, depending on population density and agricultural production intensity. Without having a spatially-differentiated life cycle inventory for the polypropylene liner, or a spatially-differentiated impact assessment model for geographic regions outside Western Europe and North America, it would be difficult to determine if the assessment of the Remediation Activities' impact could be mitigated by spatial differentiation.

7.5 Uncertainty and data requirements

The parameter uncertainty associated with the MNA and IMPACT models stress the need for high quality primary data in making environmental comparisons, such as what has been carried out with the present study. In particular, partitioning coefficients and decay rates have been identified as key parameters for modeling chemical fate, but which can introduce significant uncertainty to model results.

In order to demonstrate the feasibility of MNA as a site management option, the bioavailability of contaminants and the transport possibilities must be assessed, as well as attenuation processes including decay rates and other physico-chemical phenomena. The present study has been carried out with the assumption that the parameter values used are taken from such a feasibility study.

7.6 Ecosphere and technosphere perspectives

LCA of contaminated site remediation projects can consider the site as part of the ecosphere or part of the technosphere, depending on the goals and scope of the LCA. Not surprisingly, the present study showed that the results can change depending on the perspective taken. For the MWW management project, this perspective only had an influence on the Ecosystem Quality damage impact category and the midpoint category of Aquatic Ecotoxicity, as it was assumed that the contaminated sediments did not contribute to Human Health impacts directly.

The choice between the ecosphere and technosphere perspective influences the overlap that the LCA will have with a risk assessment carried out for the site. The choice to consider the site as part of the technosphere creates a clearer distinction between impacts that will be evaluated using LCA and those that will be evaluating using a risk assessment. It is possible to take this distinction one step further and not evaluate any contaminant-related impacts within the LCA framework, rely solely on risk assessment results for this evaluation and leave LCA to consider only secondary impacts. This choice may simplify decision-making in some cases and avoids the difficulties associated with evaluating primary impacts using LCIA, however discounts the possibility for an important comparison of primary and secondary impacts.

All other factors being equal, the decision to treat the contaminated site as part of the technosphere seems most appropriate when exposure to on-site contamination is limited. Such cases would include certain contaminated soil remediation projects, where an LCIA evaluation of the on-site contamination would lead to an overestimation of primary impacts, as has been found in previous remediation LCA studies (Godin *et al.*, 2004; Dontigny, 2004; Toffoletto *et al.*, 2005; Lesage *et al.*, 2007; Cadotte *et al.*, 2007).

A final but important consideration for deciding what perspective to take will be the confidence of stakeholders in the indicators used to assess primary impact. For the present study, given that direct exposure of benthic organisms to the on-site contamination exists and represents a significant environmental impact, it seems suitable to consider the site as part of the ecosphere. Should the project stakeholders have confidence in the sediment ecotoxicity effect factors, then an ecosphere perspective is advocated.

7.7 Comparison of primary and secondary impacts with LCIA

In addition to other emissions, the Remediation Activities were responsible for five million kilograms of nitrogen oxides and sulfur dioxide emitted into air as well as 36,700 kilograms of aromatic hydrocarbons. It is not inconceivable that this could cause greater Human Health and Ecosystem Quality impacts than approximately 325,000 kilograms of metals and 10,000 kilograms of organics located in the deep and mixed sediments of MWW and their resulting emissions. However, the conclusion that MNA causes the same or less impact than the Remediation Activities for all damage impact categories is a sensitive result to interpret. It is unlikely that stakeholders would accept this evaluation, or the conclusion that the site should not be remediated.

Such a result highlights the divergence of LCA from risk assessment. LCIA's assumptions of linearity and additivity in cause-and-effect chains, the disregard for thresholds and the consideration of impacts to all species rather than simply the most sensitive could easily result in divergent conclusions between the two analyses. Add to this the effect of methodological weaknesses and inherent uncertainty, notably in the treatment of metals, and the possibility for divergence increases.

It should be noted that there is no need for convergence of the two analyses and that divergent results speak to LCA's ability to account for important factors that risk assessment can not evaluate.

It can not be repeated enough that LCIA is not intended to evaluate real impacts, but is meant to produce indicators with comparative value only. That said, scientific development behind LCIA has been focussed on improving the "environmental realism" of these indicators, such that they better reflect actual environmental insults. The present study sought to improve LCIA toxicity and ecotoxicty indicators through site-specific fate modeling to increase the confidence in a comparison of primary and secondary impacts. Despite these efforts, it seems probable that a subjective weighting system would be required to assist in such a comparison, likely weighting primary impacts heavier than secondary.

8 Conclusions and Recommendations

The generic fashion in which LCIA models evaluate emissions can make them inappropriate for assessing important site-specific impacts, such as those associated with contaminated sites. This master's thesis investigated the effect of integrating site specificity into the LCA of a remediation project to manage a coastal industrial site with contaminated sediments. The following hypothesis was investigated:

The conclusions of a life cycle assessment to evaluate contaminated site remediation options can be changed according to the level of spatial differentiation integrated within the life cycle impact assessment method.

The spatially differentiated fate and exposure model IMPACT was used to integrate site-specific data into the LCIA of water and air emissions from the contaminated site. Water emissions were evaluated with a non-spatially differentiated model, a spatially differentiated model using watersheds as the level of resolution, as well as a modified version of this latter model with an additional zone representing the receiving body of water on which the contaminated site is located. Air emissions were evaluated with a non-spatially differentiated model and a spatially differentiated model that considers an urban environment nested within the watershed spatial plan. Impacts due to on-site sediment contamination were evaluated using sediment ecotoxicty effect factors, calculated using aquatic ecotoxicity effect factors and sediment-water partitioning coefficients according to the Equilibrium Partitioning method.

The characterization factors produced by these models were used to evaluate the impact caused by emissions from the contaminated site. These emissions were calculated using a mass balance model that predicted the concentrations of contaminant in the sediments and overlying water column. Analysis of model results showed advection to be a dominant process in removing contaminants from the water column. Removal from the mixed sediments was best described by decay, with the fate of the remaining contaminants being split between resuspension and burial, according to the velocities of these processes.

The integration of site-specific data into IMPACT produced differences in the characterization factors used to evaluate water emissions due to differences in water residence times experienced by an emitted contaminant. Differences in the ecotoxicity characterization factors used to evaluate air emissions were attributed to differences in rainfall rate, runoff rate and the relative sizes of the soil and water compartments. Agricultural output and the use of the emission zone body of water as a drinking source were responsible for differences between human toxicity characterization factors used to evaluate air emissions.

The characterization factors produced by models with different levels of spatial differentiation resulted in different LCA conclusions. Off-site emissions attributed to the monitored natural attenuation option caused less impact than dredging and confined disposal in the Human Health and Ecosystem Quality impact categories when the receiving body of water was modeled in developing human toxicity and ecotoxicity characterization factors. These results affirmed the hypothesis stated above.

The following recommendations can come out of the present study:

Mass balance models used to describe contaminated sites will be sensitive to parameters that show significant natural variation. Site conditions must be well studied and mechanisms governing the environmental fate of contaminants should be well understood to minimize uncertainty in model results.

Sediment ecotoxicity effect factors calculated using the Equilibrium Partitioning method should be used with caution. When applying these effect factors in cases where sediment impacts are important, the mechanisms governing sediment ecotoxicity should be well understood. Specifically, an investigation of the most important exposure routes should be carried out, as well as whether hydrophobicity mitigates toxicity.

Sediment ecotoxicity factors were applied in the present study where a known impact existed. A more general application of sediment ecotoxicity factors should be considered if a cause-effect relationship between sediment concentrations and ecotoxic impact can be demonstrated and described in a manner compatible with LCIA methods.

Where feasible, spatial differentiation in LCIA methods that allows for the integration of site-specific parameter values should be performed when assessing important impacts within a system's life cycle, if a spatially-differentiated inventory is available. Access to accurate data on water residence time and agricultural output of the relevant region were shown to be important for the contaminants studied here. Accurate fate modeling of metals should be prioritized in studies where metal emissions are important.

Further research into the fate modeling of metals is required to account for speciation and bioavailability. Factors that govern the residence time of metals in aquatic environments should be included in these fate modeling efforts.

Caution is warranted in the application of LCIA in evaluating primary impacts associated with site remediation projects. The evaluation of impacts caused by contaminants with little risk of exposure may be inappropriate. Confidence in life cycle impact indicators will increase if the appropriate perspective is taken when considering the contaminated site as part of the ecosphere or technosphere.

With respect to the MWW contaminated site case study the following recommendations can be made:

Alternative management options that may have lesser secondary impacts should be considered for this site. The mass of polypropylene liner should be minimized as much as is feasible or an alternate site found that does not require a liner. If possible, a disposal site closer to MWW should be found.

The site should be evaluated as part of the ecosphere, as benthic species are exposed to the sediment contamination, causing considerable ecotoxic impacts.

The use of LCIA to evaluate primary impacts seems warranted for the MWW site. While the impacts measured are only indicators, they provide a common scale with which to evaluate the impacts caused by the management alternatives. The consideration of alternative remediation scenarios might not appropriately weigh secondary impacts, were it not for their magnitude with respect to the primary impacts produced by the on-site contamination.

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APPENDICES

Appendix 1. IMPACT models

Table A1.1. Parameters adaptable to specific geographic regions (Humbert *et al.*, 2009)

Watershed zone modeling		
Soil module:	***Bulk vegetation module:***	***Water module:***
rainfall infiltration fraction (-) runoff rate (fraction of rainfall) ppm of solids in runoff water (ppm) solid phase soil density (kg/m3) fraction organic carbon in solid phase of soil (-) area soil (m2) temp (K) air boundary layer thickness (m) rainfall rate (m/h) ***Surface layer:*** depth (m) volumetric water fraction (-) volumetric air fraction (-) volumetric solids fraction (-) ***Agricultural root zone:*** depth (m) volumetric water fraction (-) volumetric air fraction (-) volumetric solids fraction (-) ***Vadose layer:*** depth (m) volumetric water fraction (-) volumetric air fraction (-) volumetric solids fraction (-)	dry plant mass (kg/m2-soil) volume fraction leaf to aerial plant part (-) volume fraction root to aerial plant part (-) transpiration coefficient (l /kg) leaf area index (-) type of land cover (-) diffusion length stomata (m) diffusion length boundary layer (m) density root (bulk) (kg/l)correction plant lipid-n-octanol (barley) (-) root water content (g/g) root lipid content (g/g) density stem (bulk) (kg/l) correction plant lipid-n-octanol (barley) (-) stem water content (g/g) stem lipid content (g/g) density leaf (bulk) (g/g) correction plant lipid-n-octanol (barley) (-) leaves water content(g/g) leaves lipid content (g/g) temp (K) fraction of area soil covered by agri. vegetation (g)	mean depth (m) surface area (m2) water pH (-) temperature (annual average) (K) diffusion constant of oxygen in water at the temperature T (cm2/s) diffusion constant of water in air at the temperature T (cm2/s) mass fraction of organic matter in suspended solids (-) ***Sediment-land module:*** sediment to water mass transfer coefficient (m/h) fraction of sediment deposition buried (-) fraction of sediment deposition resuspended (-) solid phase sediment density (kg/m3) fraction of organic carbon in solid phase of sediment (-) mean sediment accumulation sediment depth (m) area sediment (m2) solids volume fraction(-)

Table A1.1 (cont'd). List of parameters adaptable to specific geographic regions (Humbert *et al.*, 2009)

Watershed zone modeling		
Exposure module: population (number of persons) ***usable production:*** sum unexposed produce (kg/yr) sum exposed produce (kg/yr) fresh water fish (kg/yr) pigs (kg/yr)	***Exposure module:*** beef+veal (kg/yr) broilers (kg/yr) goat and sheep (kg/yr) eggs (kg/yr) dairy products (cow milk) (kg/yr) fraction of surface drinking water (-)	***Number of head:*** pigs (nb) beef+veal (nb) broilers (nb of utility chicks) goat and sheep (nb) laying hens (nb) dairy cattle (nb)

Air zone modeling		
Air module: dry deposition velocity (m/h) wet deposition scavenging volume (m^3 air/m^3 rain) air boundary layer thickness (m) rainfall rate (m/h) average dry period (hours) lower to upper atmospheric loss rate const. (m/h) temp (K) aerosol solid density (kg/m3) height of lower atmospheric boundary layer (m) aerosol phase fraction (m3/m3 air) paved Area with drainage systems (m2)	***Air module:*** area water (m2) area soil (m2) fraction of area soil covered by agri. vegetation (-) capture coefficient (klai) leaf area index (-) diffusion length stomata (m) diffusion length stomatal boundary layer (m) ***Surface soil layer module:*** volumetric water fraction (-) volumetric air fraction (-) volumetric solids fraction (-) ***Exposure module:*** population (number of persons)	***Usable production in considered zone:*** sum unexposed produce (kg/yr) sum exposed produce (kg/yr) pigs (kg/yr) beef (kg/yr) broilers (kg/yr) goat and Sheep meat (kg/yr) eggs (kg/yr) dairy products (cow milk) (kg/yr) ***Number of head:*** pigs (nb) beef+veal (nb) broilers (nb of utility chicks of table strains hatched) goat and sheep (nb) laying hens (nb) dairy cattle (nb)

Table A1.1 (cont'd). List of parameters adaptable to specific geographic regions (Humbert *et al.*, 2009)

Coastal/ocean zone modeling		
Oceanic water module:	***Sediment module:***	***Sediment module:***
depth of upper layer (m)	depth of sediment (m)	fraction of sediment deposition buried (-)
depth of lower layer (m)	area sediment (m2)	fraction of sediment deposition resuspended (-)
water pH (-)	solids fraction (-)	***Exposure module (usable production in considered zone):***
suspended sediment density (kg/m3)	solid phase sediment density (kg/m3)	sea fish (kg/yr)
fraction of organic carbon in suspended sediment (-)	fraction of organic carbon in solid phase of sediment (-)	
vertical mixing velocity (m/h)	sediment to water mass transfer coefficient (m/h)	
area oceanic water (m2)	sediment deposition rate constant (m/h)	
temp (K)		
sediment deposition rate constant (m/h)		
sediment to water mass transfer coefficient (m/h)		

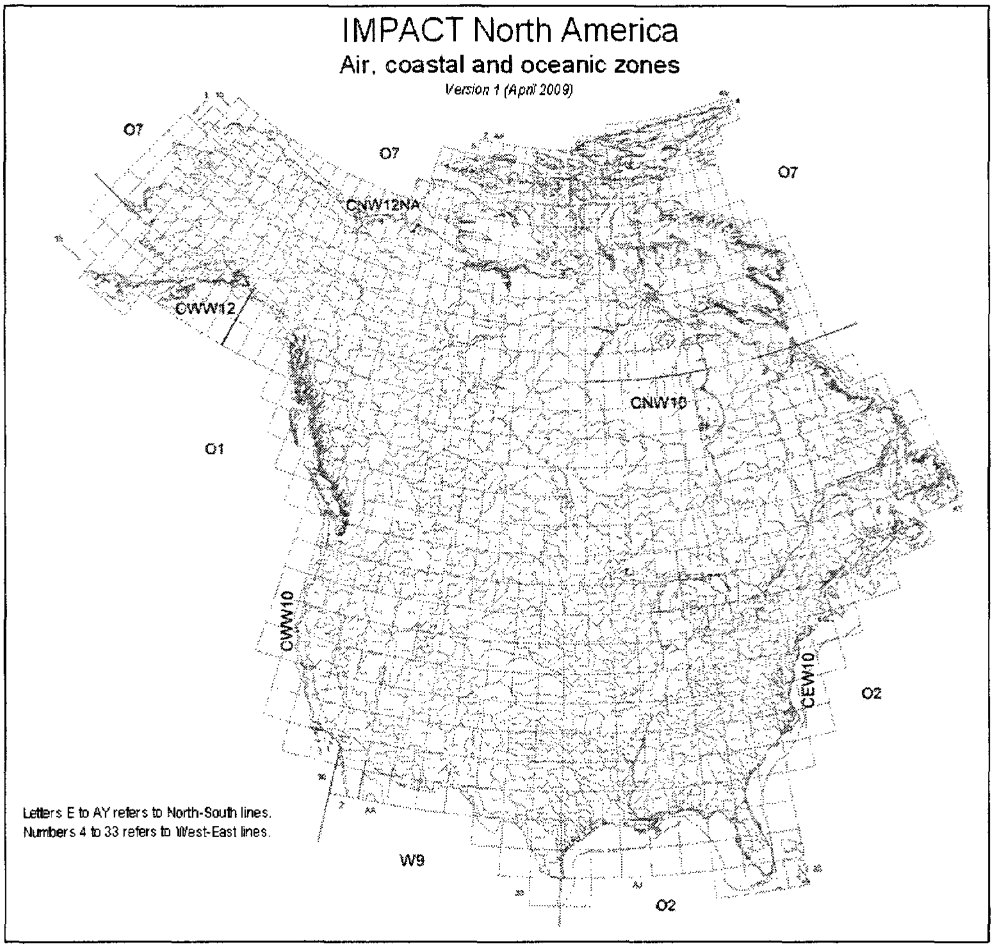

Figure A1.1. Spatial layout of airsheds, watersheds and coastal zones modeled by IMPACT North America. Figure taken from Humbert *et al.* (2009).

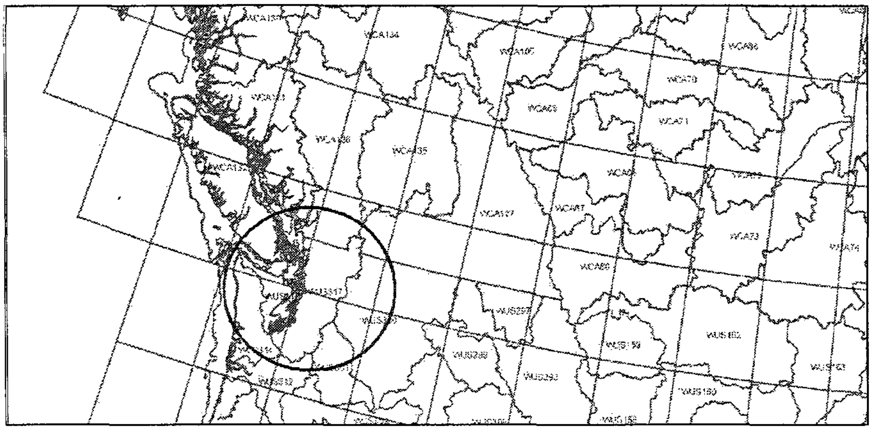

Figure A1.2. Location of zone WUS317, representing HUC 171100.

Table A1.2. Watershed parameters for the emission zones of the Non-spatial, Modified Watershed (Puget Sound) and Watershed (WUS317) models, including zones downwind to zone WUS317.

	Emission zones		
Parameter	**Non-spatial Canada**	**Puget Sound**	**WUS317**
Area soil [m^2]	9.42E+12	1.00E+00	3.71E+10
Rainfall rate [m/h]	7.44E-05	1.11E-04	2.13E-04
Runoff rate [fraction of rainfall]	0.067	1.000	0.366
Mean depth [m]	15.1	62.5	5.0
Surface area [m^2]	1.29E+12	2.64E+09	2.83E+09
Area sediment [m^2]	1.29E+12	2.64E+09	2.83E+09
Population	3.00E+07	0.00E+00	4.00E+06
Unexposed produce [kg/yr]	1.84E+10	0.00E+00	3.42E+06
Exposed produce [kg/yr]	5.05E+10	0.00E+00	5.06E+06
Fresh water fish [kg/yr]	1.08E+08	3.68E+07	3.68E+07
Pigs [kg/yr]	1.40E+09	0.00E+00	1.88E+03
Beef [kg/yr]	4.06E+09	0.00E+00	1.88E+05
Broilers [kg/yr]	1.17E+09	0.00E+00	4.33E+04
Goat and sheep meat [kg/yr]	2.26E+07	0.00E+00	1.12E+03
Eggs [kg/yr]	5.09E+08	0.00E+00	5.11E+04
Dairy products [kg/yr]	4.77E+08	0.00E+00	8.02E+05
Fraction of surface drinking water	0.77	0.0	0.77

Table A1.2 (cont'd). Watershed parameters for the emission zones of the Non-spatial, Modified Watershed (Puget Sound) and Watershed (WUS317) models, including zones downwind to zone WUS317.

	Downwind to WUS317			
Parameter	**WUS300**	**WUS301**	**WUS312**	**WUS314**
Area soil [m^2]	5.76E+10	1.59E+10	1.59E+10	1.51E+10
Rainfall rate [m/h]	6.16E-05	8.12E-05	2.34E-04	3.05E-04
Runoff rate [fraction of rainfall]	0.366	0.366	0.366	0.366
Mean depth [m]	5.0	5.0	5.0	5.0
Surface area [m^2]	4.62E+09	1.25E+09	1.06E+09	1.19E+09
Area sediment [m^2]	4.62E+09	1.25E+09	1.06E+09	1.19E+09
Population	4.07E+05	3.32E+05	5.92E+05	1.80E+05
Unexposed produce [kg/yr]	1.57E+09	4.23E+08	3.49E+08	4.03E+08
Exposed produce [kg/yr]	2.32E+09	6.26E+08	5.15E+08	5.96E+08
Fresh water fish [kg/yr]	1.81E+06	4.90E+05	3.83E+05	4.67E+05
Pigs [kg/yr]	8.62E+05	2.33E+05	2.05E+05	2.22E+05
Beef [kg/yr]	8.63E+07	2.33E+07	2.33E+07	2.22E+07
Broilers [kg/yr]	1.98E+07	5.35E+06	4.58E+06	5.10E+06
Goat and sheep meat [kg/yr]	5.11E+05	1.38E+05	1.85E+05	1.31E+05
Eggs [kg/yr]	2.34E+07	6.32E+06	5.46E+06	6.02E+06
Dairy products [kg/yr]	3.67E+08	9.92E+07	8.41E+07	9.45E+07
Fraction of surface drinking water	0.77	0.77	0.77	0.77

Table A1.2 (cont'd). Watershed parameters for the emission zones of the Non-spatial, Modified Watershed (Puget Sound) and Watershed (WUS317) models, including zones downwind to zone WUS317.

	Downwind to WUS317		
Parameter	**WCA135**	**WCA136**	**WCA137**
Area soil [m^2]	5.43E+10	6.19E+10	1.03E+11
Rainfall rate [m/h]	5.41E-05	1.69E-04	7.56E-05
Runoff rate [fraction of rainfall]	0.366	0.366	0.366
Mean depth [m]	8.8	5.0	7.7
Surface area [m^2]	1.45E+09	1.69E+09	4.41E+09
Area sediment [m^2]	1.45E+09	1.69E+09	4.41E+09
Population	1.82E+05	1.81E+06	4.65E+05
Unexposed produce [kg/yr]	9.61E+06	1.67E+07	8.24E+07
Exposed produce [kg/yr]	1.16E+07	2.27E+07	3.67E+08
Fresh water fish [kg/yr]	3.25E+06	3.80E+06	9.91E+06
Pigs [kg/yr]	1.41E+06	1.02E+07	1.24E+06
Beef [kg/yr]	2.89E+07	5.88E+07	1.03E+08
Broilers [kg/yr]	6.11E+06	4.71E+07	1.73E+07
Goat and sheep meat [kg/yr]	1.96E+05	3.09E+05	4.65E+05
Eggs [kg/yr]	1.27E+06	9.81E+06	3.59E+06
Dairy products [kg/yr]	5.13E+06	1.04E+07	1.83E+07
Fraction of surface drinking water	0.77	0.77	0.77

Table A1.3. Urban box model parameters for the City of Tacoma, Washington, USA.

City	**Air cell**	**Watershed**	**Urban population**	**Source**	**Area (km^2)**	**Source**	**Density ($pers/km^2$)**
Tacoma	X22	WUS317	196,520	Wikipedia (2009)	162.2	Wikipedia (2009)	1211.6

Table A1.4. Chemical parameters used in the IMPACT models for the contaminants at MWW site.

	Henry's constant [Pa m^3 mol^{-1}]	**Log Kow**	**Degradation half life in air [h]**	**Degradation half life in water [h]**	**BCF (kg-water/kg-fish)**
1,2-Dichlorobenzene	192	3.4	5.50E+02	1.70E+03	87.32
2,4-Dimethylphenol	0.095	2.35	1.70E+01	5.50E+01	11.78
Acenaphthene	18.4	3.92	5.50E+01	5.50E+02	208.2
Anthracene	5.56	4.54	5.50E+01	5.50E+02	532.7
Benzo(a)anthracene	1.2	5.91	1.70E+02	1.70E+03	5435
Benzo(a)pyrene	0.046	6.04	1.70E+02	1.70E+03	10470
Benzo(b)fluoranthene	0.066	5.78	2.10E+01	3.65E+02	5631
Butyl benzyl phthalate	0.126	4.68	5.50E+01	1.70E+02	875.2
Chrysene	0.523	5.61	1.70E+02	1.70E+03	5939
DDD	0.66	6.02	8.86E+01	7.88E+04	8618
DDE	4.16	5.7	1.70E+02	5.50E+04	20540
DDT	0.832	6.19	1.70E+02	5.50E+03	41750
Dibenz(a,h)anthracene	0.014	6.75	7.70E+00	3.94E+02	21670
Di-n-butyl phthalate	0.181	4.72	5.50E+01	1.70E+02	582.1
Di-n-octyl phthalate	0.257	8.1	1.87E+01	3.36E+02	63.53
Fluoranthene	0.886	5.22	1.70E+02	1.70E+03	1876
Fluorene	9.62	4.18	5.50E+01	5.50E+02	330.1
Indeno(1,2,3-c,d)pyrene	0.035	6.7	6.00E+00	1.59E+04	28620
Naphthalene	44	3.37	1.70E+01	1.70E+02	69.34
N-Nitrosodiphenylamine	75.29	3.13	5.00E+00	1.70E+01	51.3
PCB (total)	33.6	7.55	1.90E+03	1.44E+03	4922
Pentachlorophenol	0.00	5.05	5.50E+02	5.50E+02	695.7
Phenol	0.03	1.46	1.25E+01	2.24E+01	2.656
Pyrene	1.19	5.18	1.70E+02	1.70E+03	1142

Henry's constant for mercury is 261.3 Pa-m^3-mol^{-1}.

Table A1.4 (cont'd). Chemical parameters used in the IMPACT models for the contaminants at MWW site.

	Median estimates of ED10 [mg / kg body weight – day]				**Aquatic Ecotoxicological Effect Factor [PAF per kg/m3]**
	Oral - non cancer	**Inhalation - non cancer**	**Oral - cancer**	**Inhalation - cancer**	
1,2-Dichlorobenzene	7.14E+00	7.14E+00			6.16E+02
2,4-Dimethylphenol	1.85E+00	1.85E+00			7.28E+01
Acenaphthene	6.67E+00	6.67E+00			4.15E+03
Anthracene	3.70E+01	3.70E+01			3.92E+04
Benzo(a)anthracene	3.70E-02	3.70E-02			2.45E+05
Benzo(a)pyrene			3.85E-02	3.85E-02	2.27E+04
Benzo(b)fluoranthene	3.70E-02	3.70E-02			2.31E+05
Butyl benzyl phthalate	1.28E+01	1.28E+01			2.27E+03
Chrysene	3.70E-02	3.70E-02			2.74E+05
DDD			1.23E+00	1.23E+00	1.38E+05
DDE			6.67E-01	6.67E-01	2.88E+04
DDT	1.33E-02	1.33E-02	3.33E+00	5.88E-01	1.48E+05
Dibenz(a,h)anthracene			2.33E-01	2.33E-01	1.25E+07
Di-n-butyl phthalate	1.01E+01	1.01E+01			2.85E+03
Di-n-octyl phthalate	3.09E+00	3.09E+00			1.50E+01
Fluoranthene	4.76E+00	4.76E+00			1.22E+03
Fluorene	4.76E+00	4.76E+00			2.60E+03
Indeno(1,2,3-c,d)pyrene	3.70E-02	3.70E-02			1.70E+06
Naphthalene	5.88E+00	5.88E+00	6.67E+00	6.67E+00	2.05E+03
N-Nitrosodiphenylamine			6.67E+00	6.67E+00	
PCB (total)			7.14E-02	7.14E-02	7.50E+05
Pentachlorophenol	7.69E-01	7.69E-01	1.64E+00	1.64E+00	9.01E+03
Phenol	1.59E+01	1.59E+01			8.26E+01
Pyrene	2.78E+00	2.78E+00			2.56E+04
Antimony	1.75E-02	1.75E-02			9.00E+02
Arsenic	1.28E-03	1.28E-03	1.33E-01	1.33E-02	2.20E+03
Cadmium	7.50E-03	7.50E-03	1.64E-02	1.64E-02	2.62E+02
Copper	2.27E+00	2.27E+00			1.47E+04
Lead	2.25E-01	2.25E-01			1.16E+03
Mercury	1.74E-02	7.71E-04			2.31E+04
Nickel	1.25E+00	1.29E-02			5.78E+02
Zinc	9.09E-02				1.52E+03

Appendix 2. Management Scenario Technical Details

Option 1: Monitored Natural Attenuation

Calculation details are given here for a sediment monitoring program of 50 years in length. One sampling round is performed each year at 30 sampling locations. Two samples are taken from each sampling location using a gravity core sampler (Miller & Dorkin, 2001). Calculated and assumed values used for determining the LCI of the 50-year sampling campaign are given in Table A2.1 below. The table has been laid out to reflect the steps required for calculating the inventory.

Table A2.1. LCI calculations for Monitored Natural Attenuation

Parameter	**Unit**	**Value**	**Source**
Monitoring Program Length	year	50	assumed
Sampling rounds per year	-	1	assumed
Sampling locations	-	20	assumed
Samples/location	-	2	assumed
Core volume	litre	2	Miller & Dorkin (2001)
Sediment sample density	kg/m^3	1600	Chapra (1997)
Sample mass	kg	3.2	calculated
50L Storage cooler mass + ice	kg	12.9	Coleman.com
Storage coolers/sampling round	-	2	calculated
Sampling cycle time	hours/sample	0.5	assumed
Boat operation per sampling event	hours	20	calculated
Total Boat Use	hours	1000	calculated
Engine Power	HP	10	Nissan (2009)
Normalized fuel consumption	kg/h-HP	0.227	Nissan (2009)
Hourly fuel consumption	kg/h	2	calculated
Sampling Boat Diesel Engine Emission Factors			
Hydrocarbons	g/kW-hr	0.223	USEPA (2002a)*
Nitrogen oxides (NOx)	g/kW-hr	6.77	USEPA (2002a)
Carbon dioxide	g/kW-hr	724.13	USEPA (2002a)
Carbon monoxide	g/kW-hr	0.961	USEPA (2002a)
Particulate matter	g/kW-hr	0.166	USEPA (2002a)

Table A2.1 (cont'd). LCI calculations for Monitored Natural Attenuation

Parameter	**Unit**	**Value**	**Source**
Transport			
Transport distance	km	30	assumed
Transported per year	ton-km/year	4.61	calculated
LCI 50-Year Sampling Campaign			
Diesel fuel consumption	kg	2270	calculated
Hydrocarbon emissions	kg	2.23	calculated
Nitrogen oxide emissions	kg	67.7	calculated
Carbon dioxide emissions	kg	7241.3	calculated
Carbon monoxide emissions	kg	9.6	calculated
Particulate matter emissions	kg	1.7	calculated
Transport	ton-km	230.6	calculated

*Emission factors for marine diesel engines in the 2008 version of Nonroad have not been changed since the 2002 update (USEPA, 2008).

Option 2: Dredging with disposal in a confined disposal facility

Calculation details for dredging and confined disposal are given in Table A3.2 The table has been laid out to reflect the steps required for calculating the inventory.

Table A2.2 LCI calculations for Dredging and Confined Disposal

Parameter	**Unit**	**Value**	**Source**
Dredging Operations			
Volume sediments	m^3	83719	Anchor & Foster Wheeler (2001)
Volume dredged with DB Viking Derrick	m^3	41859	assumed
Volume clamshell bucket	m^3	9.17	Godin (2004)
Load factor	-	0.75	assumed
Cycles required with Derrick	-	6083	calculated
Derrick cycle time	h	0.05	assumed
Derrick use	h	304	calculated
Derrick power	HP	350	JJM (2008), Terex (2009)
Derrick energy use	HP-h	106458	calculated

Table A2.2 (cont'd). LCI calculations for Dredging and Confined Disposal

Parameter	**Unit**	**Value**	**Source**
Dredging Operations			
Volume dredged with 6" hydraulic suction dredge	m^3	41859	assumed
Suction dredge production	m^3/h	31.8	Herbich (2000)
Suction dredge use	h	1316	calculated
Suction dredge power	HP	5	Herbich (2000)
Suction dredge energy use	HP-h	6582	calculated
Sediment Transport-Truck (16-32 t)	t-km	5023125	calculated
Distance transported	km	40	assumed
Density wet sediments	t/m^3	1.5	Herbich (2000)
Mass sediments	t	125578	calculated
Backfilling*			
Volume backfill: gravel with sand	m^3	8372	Anchor & Foster Wheeler (2001)
Density	t/m^3	1.92	Herbich (2000)
Mass backfill	t	16074	calculated
Distance transported	km	50	assumed
Backfill transport-Truck (16-32 t)	t-km	803700	calculated
Backfill placement-front end loader Cat 980g			
Bucket capacity	m^3	5.35	Godin (2004)
Loader cycles	-	1564	calculated
Loader cycle time	h	0.05	assumed
Loader use	h	78	calculated
Loader power	HP	311	Godin (2004)
Loader energy use	HP-h	24325	calculated

*Backfilling to the original grade is required by the USEPA (2000) to prevent habitat loss. Backfill is placed in areas where the original grade was 10 feet below the MLLW mark and above, which makes up approximately 10% of the MWW site dredged area.

Table A2.2 (cont'd). LCI calculations for Dredging and Confined Disposal

Parameter	Unit	Value	Source
CDF Construction			
CDF area (200m x 200m)	m^2	40000	assumed
CDF dike wall length	m	200	calculated
Required dike height	m	2.5	USEPA (1993)
Dike width (3:1 slope)	m	7.5	USEPA (1993)
Volume clay required for dike	m^3	7500	calculated
Clay density	t/m^3	1.09	SI Metric (2008)
Clay mass	t	8168	calculated
Distance clay transported	km	50	assumed
Clay transport-Truck (16-32t)	t-km	408375	calculated
PVC perimeter piping length	m	800	calculated
6" Sched 80 PVC density	kg/m	0.38	Engineering Toolbox (2005)
Mass PVC piping	t	0.304	calculated
Distance PVC piping transport	km	15	assumed
PVC piping transport-Truck (16-32t)	t-km	4.57	calculated
HDPE Liner area	m2	40000	assumed
Liner thickness	m	0.07	USEPA (1994)
Liner density	kg/m2	0.339	CETCO (2009)
HDPE liner mass	t	13.56	calculated
Distance liner transported	km	50	assumed
Liner transport-Truck (16-32t)	t-km	678	calculated
Gravel	t	4084	USEPA (1993)
Distance gravel transported	km	50	assumed
Gravel transport-Truck (16-32t)	t-km	204188	calculated
Site preparation			
Ground clearing 0.75m depth - dozer	m^3	30000	assumed/calculated
Cat D7R Series II dozer production	m^3/h	300	Godin (2004)
Dozer use	h	100	calculated
Dozer power	HP	240	Godin (2004)
Dozer energy use - clearing	HP-h	24000	calculated
Construction - piling Cat D7R Series II Dozer	m^3	7500	calculated
Dozer use	h	25	calculated
Dozer energy use - piling	HP-h	6000	calculated

Table A2.2 (cont'd). LCI calculations for Dredging and Confined Disposal

Parameter	Unit	Value	Source
CDF Construction			
Profiling - excavator - Cat 320C L	m^3	7500	calculated
Cat 320C L Excavator production	m^3/h	313	Godin (2004)
Excavator use	h	24	calculated
Excavator power	HP	138	Godin (2004)
Excavator energy use	HP-h	3302	calculated
Compacting - Vibratory roller Cat CS-563D	m^2	6325	calculated
Cat CS-563D roller production	m^2/h	60	Godin (2004)
Roller use	h	105	calculated
Roller power	HP	145	Godin (2004)
Roller energy use	HP-h	15284	calculated
CDF Closure			
Sediment compaction - Vibratory roller: Cat CS-563D	m^2	40000	calculated
Cat CS-563D roller production	m^2/h	60	Godin (2004)
Roller use	h	667	calculated
Roller power	HP	145	Godin (2004)
Roller energy use	HP-h	96667	calculated
Grading - dozer (assume 0.3 m depth)	m^3	12000	calculated
Cat D7R Series II dozer production	m^3/h	300	Godin (2004)
Dozer use	h	40	calculated
Dozer power	HP	240	Godin (2004)
Dozer energy use - grading	HP-h	9600	calculated
CDF 30cm Cap - Sand	m3	12000	calculated
Sand density	t/m^3	1.5	SI Metric (2008)
Mass sand cap	t	18000	calculated
Distance sand transported	km	50	assumed
Cap placement - Cat D7R Series II Dozer	m^3	12000	calculated
Cat D7R Series II dozer production	m^3/h	300	Godin (2004)
Dozer use	h	40	calculated
Dozer power	HP	240	Godin (2004)
Dozer energy use - grading	HP-h	9600	calculated

Table A2.2 (cont'd). LCI calculations for Dredging and Confined Disposal

Parameter	**Unit**	**Value**	**Source**
Post-dredging monitoring program			
Monitoring Program Length	year	10	assumed
Sampling rounds per year	-	1	assumed
Sampling locations	-	5	assumed
Samples/location	-	2	assumed
Core volume	litre	2	Miller & Dorkin (2001)
Sediment sample density	kg/m^3	1600	Chapra (1997)
Sample mass	kg	3.2	calculated
50L Storage cooler mass + ice	kg	12.9	Coleman.com
Storage coolers/sampling round	-	1	calculated
Sampling cycle time	h/sample	0.5	assumed
Boat operation per sampling event	h	5	calculated
Total Boat Use	h	50	calculated
Engine Power	HP	10	Nissan (2009)
Normalized fuel consumption	kg/HP-h	0.227	Nissan (2009)
Hourly fuel consumption	kg/h	2	calculated
Sampling Boat Diesel Engine Emission Factors			
Hydrocarbons	g/kW-h	0.223	USEPA (2002a)
Nitrogen oxides (NOx)	g/kW-h	6.77	USEPA (2002a)
Carbon dioxide	g/kW-h	724.13	USEPA (2002a)
Carbon monoxide	g/kW-h	0.961	USEPA (2002a)
Particulate matter	g/kW-h	0.166	USEPA (2002a)
Transport			
Transport distance	km	30	assumed
Transported per year	t-km/year	1.35	calculated
LCI 10-Year Sampling Campaign			
Sampling boat diesel fuel consumption	kg	114	calculated
Hydrocarbon emissions	kg	0.11	calculated
Nitrogen oxide emissions	kg	2.2	calculated
Carbon dioxide emissions	kg	362.065	calculated
Carbon monoxide emissions	kg	0.3	calculated
Particulate matter emissions	kg	0.1	calculated
Transport	t-km	13.5	calculated

Table A2.3 Emission factors [g/HP-h] and diesel use [g/HP-h] for dredging and construction equipment (Godin, 2004)

	DB Viking Derrick	**6" hydraulic suction dredge**	**Cat 980g loader**	**Cat D7R Series II dozer**	**Cat 320C L excavator**	**Cat CS-563D roller**
Diesel use	181.3	238.6	173.3	166.5	188.2	166.5
Particulates	0.25	0.011	0.244	0.247	0.411	0.286
NOx	5.16	2.34	3.94	5.57	6.67	5.66
VOC	0.419	0.045	0.148	0.205	0.534	0.205
NMVOC	0.42	0.061	0.138	0.192	0.499	0.192
Methane	0.016	0.001803	0.0022	0.0031	0.0081	0.0031
CO	1.81	0.9436	2.98	1.22	2.49	1.67
CO2	574.2	514.87	552.7	531	600.5	531
SO2	1.14	1.114	1.12	1.07	1.21	1.07

NO_x :	nitrogen oxides
VOC :	volatile organic compounds
NMVOC :	non-methane volatile organic compounds
CO :	carbon monoxide
CO_2 :	carbon dioxide
SO_2 :	sulfur dioxide

Appendix 3. Return Flow Factor

Calculating the water flow in Middle Waterway requires tidal data and an estimate of the quantity of water that returns to the waterway with each tide.

The fraction of water that returns to a tidal embayment after flowing out into the receiving water body is described by the return flow factor, **b**, developed by Sanford *et al.* (1992) as part of a tidal flushing model. The model combines theory from the tidal prism concept for estimating water quality in an estuary or embayment (Schwartz, 2005) with that of horizontal jet flow (Carter, 1969; cited in Sanford *et al.*, 1992). The full details of the model will not be discussed here.

Using data from nearby waterways where Middle Waterway data was not available, a return flow factor of **b** = 0.431 was calculated.

For a given tidal period, the return flow factor can be estimated using characteristic lengths associated with the tidal embayment and receiving water body. These lengths are defined according to the dimensions of the contaminant effluent plume as it leaves the embayment and is carried by the tide of the receiving water body, before being returned by the next tide. The current within the channel, ***v***, the tidal current, **U**, the initial width of the effluent plume as it leaves the channel, $\mathbf{B_o}$ and the distance of the effluent plume's axis from the shoreline, $\mathbf{y_m}$ are used with a characteristic diffusion length scale, $\boldsymbol{\sigma_T}$, to estimate the return flow factor. The characteristic diffusion length scale is defined below in Equation A3.1. The dimensions of the effluent plume are depicted below in Figure A3.1.

$$s_T = \left(0.05\$U\$h\$T\right)^{0.5} \qquad \text{(Eq. A3.1)}$$

where:

σ_T = characteristic diffusion length scale [m]

U = tidal current in the receiving body of water [m/h]

h = average depth in the receiving body of water [m]

T = tidal period [hours]

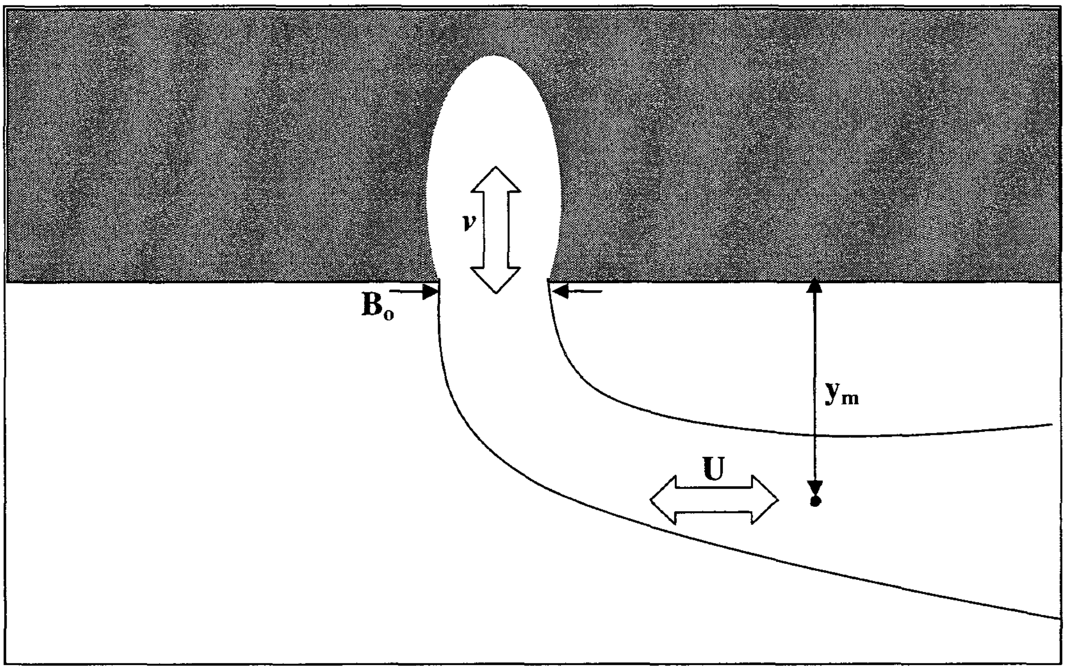

Figure A3.1. Contaminant effluent plume dimensions used in calculating the return flow factor (After Sanford *et al.*, 1992).

Sanford *et al.* (1992) demonstrated that the return flow factor can be expressed as a function of the nondimensional numbers $\mathbf{y_m/B_o}$ and $\sigma_T/\mathbf{B_o}$. The calculation of these parameters is discussed in the following paragraphs.

Data are available to calculate four of the five parameters required for the return flow factor: $\boldsymbol{v}$, **U**, $\mathbf{B_o}$, σ_T. The effluent plume's distance from the shoreline, $\mathbf{y_m}$, can be estimated using empirical relations developed by Carter (1969), relating the ratio $\mathbf{y_m/B_o}$ to the channel current to tidal current ratio, $\mathbf{r = \nu/U}$.

$$\frac{ym}{Bo} = 1.97\$r^{1.575}, \text{ for } r \text{ O } 2 \tag{Eq. A3.2a}$$

$$\frac{ym}{Bo} = 1.05\$r^{2.488}, \text{ for } 0.6 \text{ ! } r \text{ ! } 2 \tag{Eq. A3.2b}$$

$$\frac{ym}{Bo} = 0.5\$r, \text{ for } r \text{ ! } 0.6 \tag{Eq. A3.2c}$$

Tidal current data are available from the National Oceanic and Atmospheric Administration (2009a). Data are not available for Commencement Bay, however tidal currents for Gig Harbor entrance are available. The location of Gig Harbor is indicated in Figure A3.2. Channel current data for Middle Waterway are not available, but data for Hylebos Waterway are provided in Anchor & Foster Wheeler (2001).

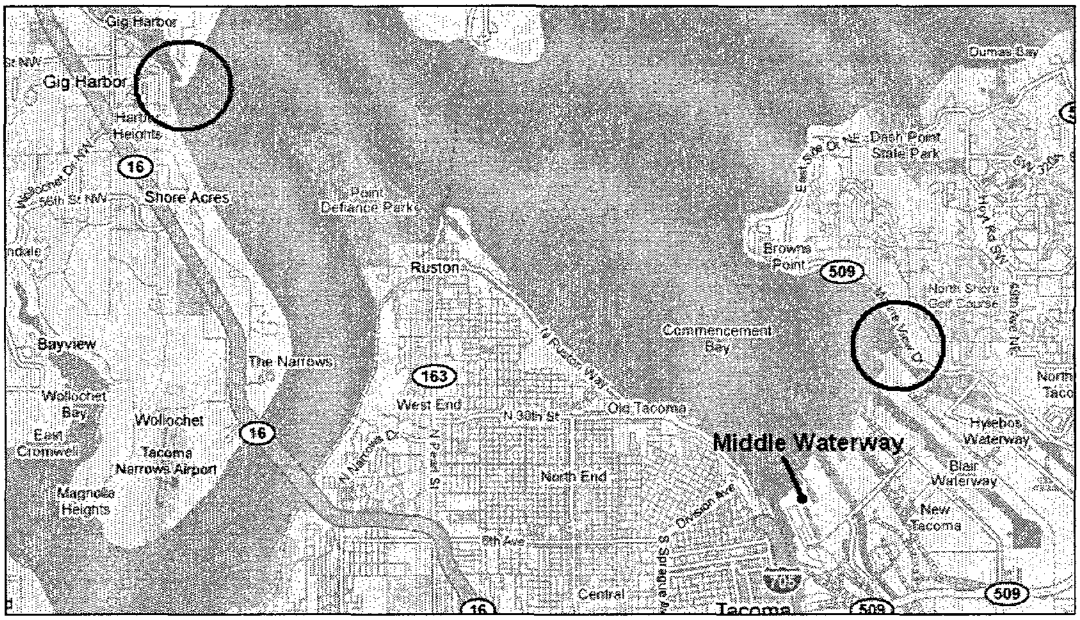

Figure A3.2. Tidal current speed for Commencement Bay will be estimated using data for Gig Harbor entrance (circled light). Channel current speed will be estimated using data from Hylebos Waterway (circled dark). Map taken from Google Maps (2009).

Parameter values used in the calculation of $\mathbf{y_m}$ and $\boldsymbol{\sigma_T}$ are given in Table A3.1. However, where the coastal current is much greater than the channel current, such that an **r**-value of less than 0.6 is obtained, a lower limit of $\mathbf{y_m} = \mathbf{B_o}/2$ is used (Sanford *et al.*, 1992). This limit indicates a modeling constraint on the effluent plume, that it maintain its original width, even as it hugs the shoreline.

Table A3.1. Parameter values used in the calculation of $\mathbf{y_m}$, distance of a contaminant effluent plume centre axis as it leaves an tidal embayment into a larger receiving body, and $\boldsymbol{\sigma_T}$, the characteristic diffusion length scale of the receiving body.

Parameter	**Unit**	**Value**
Channel current, v	m/s	0.1
Tidal current, **U**	m/s	0.463
Effluent plume width, $\mathbf{B_o}$	m	159
Effluent plume centreline distance from shoreline, $\mathbf{y_m}$	m	79.5
Average depth in Commencement Bay, **h**	m	160
Tidal period, **T**	s	44700
Characteristic diffusion length scale, $\boldsymbol{\sigma_T}$	m	2.56

Once $\mathbf{ym/B_o}$ and $\boldsymbol{\sigma_T/B_o}$ have been calculated, the return flow factor is taken from Figure A3.3, generated by successive model runs calculating the return of contaminated effluent to the channel over time. For a $\mathbf{ym/B_o}$ value of 0.5 and $\boldsymbol{\sigma_T/B_o}$ value of 2.56, the model gives a return flow factor value of **b** = 0.431.

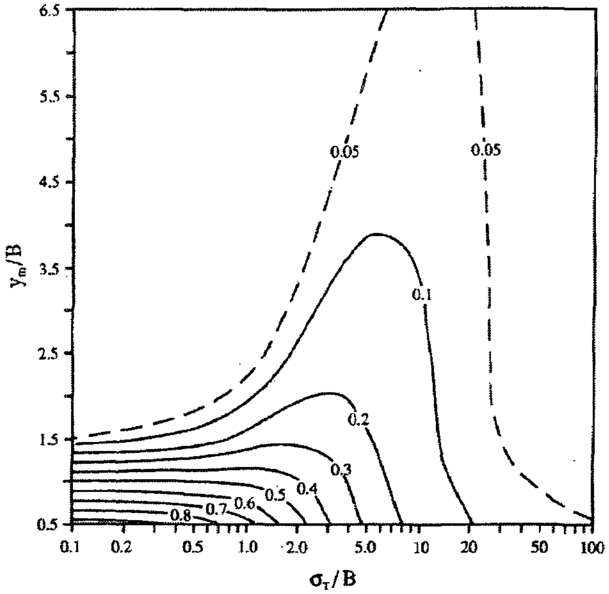

Figure A3.3. Return flow factor (**b**) as a function of the nondimensional numbers **ym/B_0** and **σ_T/B_0** (Sanford *et al.*, 1992).

The return flow factor is used to calculate the effective flow rate for Middle Waterway according to the Equation A3.3:

$$\text{effective flow rate} = \frac{\text{volume difference between tides}}{\text{tidal period}} \cdot (1 - \text{return flow factor})$$

(Eq. A3.3)

Appendix 4. Volatilization Rates

The volatilization rate, $\mathbf{k_v}$ [/day], is calculated in Equation A4.1 as:

$$k_v = F_{dw} \cdot \frac{v_v}{z_w} \qquad \text{(Eq. A4.1)}$$

where:

v_v = volatilization transfer coefficient [m/yr]

$\mathbf{z_w}$ = depth of the water column [m]

$\mathbf{F_{dw}}$ = contaminant fraction in water in dissolved form [-]

An average water column depth of 2.58 m was calculated for Middle Waterway. Calculation details for the contaminant fraction in the water column in dissolved form, $\mathbf{F_{dw}}$, are available in Appendix 6.

The volatilization transfer coefficient, v_v, can be calculated based on Whitman's two-film theory (Whitman, 1923; cited in Chapra, 1997)

$$v_v = \frac{K_l K_g H_e'}{K_g H_e' + K_l} \qquad \text{(Eq. A4.2)}$$

where:

$\mathbf{H_e'}$ = dimensionless Henry's law constant

$\mathbf{K_l}$ = liquid film mass transfer coefficient [m/yr]

$\mathbf{K_g}$ = gas film mass transfer coefficient [m/yr]

The dimensionless Henry's law constant is determined from the constant with units [Pa-m^3/mol] through the following relationship:

$$H_e' = \frac{H_e}{RT} \qquad \text{(Eq. A4.3)}$$

where:

R = the universal gas constant = 8.314 Pa-m^3/mol-K

T = temperature [K]

In cases where it was not known at what temperature the value of Henry's law constant was measured, a value of 295.65K (22.5°C) was used, as Henry's law constants reported by CalTOX and USEtox are between 20 and 25°C (McKone *et al.*, 2001; Rosenbaum *et al.*, 2008).

The liquid film mass transfer coefficient, **K$_l$** [m/yr], can be calculated according to an empirical relationship (Banks, 1975 cited in Chapra, 1997; Banks and Herrera, 1977 cited in Chapra, 1997) involving the molecular weight, **MW** [g/mol] and wind speed **U$_w$** [m/s] shown below in Equation A4.4:

$$K_l = 365 \cdot (0.728U_w^{0.5} - 0.317U_w + 0.0372U_w^2) \cdot (\frac{32}{MW})^{0.25} \qquad \text{(Eq. A4.4)}$$

An average wind speed of 3.6 m/s was taken using historical data for the Seattle-Tacoma Airport (Windfinder, 2009).

The gas film mass transfer coefficient, **K$_g$** [m/yr], can be calculated with the contaminant's molecular weight, **MW** [g/mol], according to the empirical relationship shown below in Equation A4.6 (Mills *et al.*, 1982):

$$K_g = 61320 \cdot (\frac{18}{MW})^{0.25} \qquad \text{(Eq. A4.6)}$$

Volatilization rates and parameters used in their calculation are given in Table A4.1.

Table A4.1. Volatilization rates and parameters used in their calculation

	Molecular Weight	Henry's law constant	Dissolved fraction in water column	Volatilization rate
	MW	H_e	F_{dw}	k_v
	[g/mol]	[Pa-m^3/mol]	[-]	[/day]
1,2-Dichlorobenzene	146.97	2.04E+02	0.890	2.01E+00
2,4-Dimethylphenol	122	2.03E-01	0.994	2.00E+00
Acenaphthene	154.21	4.03E+01	0.383	2.01E+00
Anthracene	178.22	4.28E+00	0.123	2.00E+00
Benzo(a)anthracene	228.29	8.13E-01	0.007	2.32E-04
Benzo(a)pyrene	252.3	9.20E-02	0.001	5.53E-06
Benzo(b)fluoranthene	252.3	1.52E+00	0.003	1.80E-04
Butyl benzyl phthalate	312	1.69E-01	0.068	5.67E-04
Chrysene	228.3	1.59E-01	0.011	8.88E-05
DDD	320.05	4.49E-01	0.005	8.94E-05
DDE	318.05	4.92E+00	0.035	2.00E+00
DDT	354.5	1.10E+01	0.010	1.49E-03
Dibenzo(a,h)anthracene	278.35	3.52E-03	0.002	2.60E-07
Di-n-butyl phthalate	278.35	1.63E-01	0.663	2.00E+00
Di-n-octyl phthalate	390.57	1.17E-01	0.005	2.55E-05
Fluoranthene	202.26	1.02E+00	0.059	5.87E-03
Fluorene	166.2	9.47E+00	0.242	2.01E+00
Indeno(1,2,3-c,d)pyrene	276.3	7.04E-03	0.000	1.25E-07
Naphthalene	128.16	4.67E+01	0.745	2.01E+00
N-Nitrosodiphenylamine	192	5.07E-01	0.722	2.00E+00
PCB (total)	340.82	1.81E+01	0.002	3.35E-04
Pentachlorophenol	266.5	2.53E-03	0.776	9.73E-05
Phenol	94	4.02E-02	0.995	6.62E-03
Pyrene	202.3	1.40E+00	0.042	4.28E-03
Antimony	122	n/a	0.874	n/a
Arsenic	74.9	n/a	0.046	n/a
Cadmium	112.41	n/a	0.000	n/a
Copper	64	n/a	0.298	n/a
Lead	207.21	n/a	0.000	n/a
Mercury	201	8.61E+02	0.015	7.86E-03
Nickel	59	n/a	0.698	n/a
Zinc	65	n/a	0.668	n/a

Appendix 5. Contaminant Partitioning

The contaminant fractions in the particulate, $\mathbf{F_{pw}}$, and dissolved phases, $\mathbf{F_{dw}}$, of the water column and in the dissolved phase of the sediments porewater, $\mathbf{F_{dp}}$, were calculated according to the formulas provided in Equations A5.1, A5.2 and A5.3, respectively (Chapra, 1997):

$$F_{pw} = \frac{K_{dw} \cdot TSS}{1 + K_{dw} \cdot TSS} \qquad \text{(Eq. A5.1)}$$

$$F_{dw} = 1 - F_{pw} = \frac{1}{1 + K_{dw} \cdot TSS} \qquad \text{(Eq. A5.2)}$$

$$F_{dp} = \frac{1}{\varphi + K_{ds} \cdot (1 - \varphi) \cdot \rho_s} \qquad \text{(Eq. A5.3)}$$

where:

K_{dw} = contaminant partitioning coefficient in water [m^3/kg]

K_{ds} = contaminant partitioning coefficient in sediments [m^3/kg]

TSS = total suspended sediment concentration [kg/m^3]

φ = porosity [-]

A TSS value of 9.2 mg/L for Middle Waterway was used for the MNA model(Anchor & Foster Wheeler, 2001), taken from an average of water quality results, measured according to the EPA Sampling Method 160.2 (USEPA, 1979).

Porosity was estimated using the percent solids measured in the mixed and deep sediments, and the density of sediment solids and seawater. An average density of 2630 kg/m^3 was used for the sediments, obtained from specific gravity measurements using the bottle pycnometer technique (Anchor & Foster Wheeler, 2001). Seawater was assumed to have a density of 1020 kg/m^3 (Anchor & Foster Wheeler, 2001). An average percent solids value of 54% was used for the mixed sediments and 55% for the deep sediments (Anchor & Foster Wheeler, 2001), giving a porosity of 0.69 for the mixed sediments and 0.68 for the deep sediments.

The contaminant partitioning coefficients for water, $\mathbf{K_{dw}}$, and for sediments, $\mathbf{K_{ds}}$, were calculated according to the formulas in Equations A5.4 and A5.5, respectively, using the organic carbon partitioning coefficient, $\mathbf{K_{oc}}$, and the fraction of organic carbon, $\mathbf{f_{oc}}$, available in each compartment.

$$K_{dw} = K_{oc} \cdot f_{oc\,(water)} \qquad \text{(Eq. A5.4)}$$

$$K_{ds} = K_{oc} \cdot f_{oc\,(sed)} \qquad \text{(Eq. A5.5)}$$

Organic carbon partitioning coefficients were taken from the CalTOX (McKone *et al.*, 2001) and USEtox (Rosenbaum *et al.*, 2008) databases. For the substances found in the Middle Waterway sediments, the majority of the partitioning coefficients taken from these databases have USEPA (1994) as their original source, except for benzo(a)pyrene (Risk Science Program, 1995). In the case of four chemicals (benzo(b)fluoranthene, DDD, di-n-octyl phthalate, indeno(1,2,3-c,d)pyrene), the organic carbon partitioning coefficient was estimated from the octanol-water partitioning coefficient, $\mathbf{K_{ow}}$, according to the formula shown below in Equation A5.6 (Karickhoff, 1981):

$$K_{oc} = 0.41 \cdot K_{ow} \qquad \text{(Eq. A5.6)}$$

Average values for the organic carbon fraction in the water column and mixed sediments were both 3.5%, with deep sediments having an average value of 2.5% organic carbon (Anchor & Foster Wheeler, 2001).

Appendix 6. IMPACT Results

Table A6.1. Intake fractions of MNA contaminants due to an emission to water for each of the three IMPACT models.

Substance	Non-spatial		Watershed		Modified Watershed	
	Oral	**Inhalation**	**Oral**	**Inhalation**	**Oral**	**Inhalation**
1,2-Dichlorobenzene	1.32E-05	2.02E-06	1.78E-05	2.07E-05	6.14E-07	1.91E-06
2,4-Dimethylphenol	5.94E-07	1.52E-10	4.92E-07	1.94E-10	2.66E-09	2.83E-11
Acenaphthene	1.13E-05	9.46E-08	1.51E-05	1.89E-07	2.16E-06	2.70E-08
Anthracene	2.45E-05	6.89E-08	5.25E-05	1.22E-07	2.58E-06	1.86E-08
Benzo(a)anthracene	6.02E-04	6.80E-08	1.11E-03	6.30E-08	3.62E-05	8.10E-09
Benzo(a)pyrene	1.23E-03	1.74E-09	1.61E-03	1.44E-09	1.92E-05	3.20E-10
Benzo(b)fluoranthene	1.71E-04	3.43E-10	2.14E-04	3.57E-10	2.09E-06	6.35E-11
Butyl benzyl phthalate	1.43E-05	9.58E-10	1.87E-05	8.12E-10	4.47E-07	1.71E-10
Chrysene	6.81E-04	4.27E-08	1.10E-03	3.95E-08	3.33E-05	6.27E-09
DDD	3.56E-03	9.60E-08	6.73E-03	1.12E-07	8.31E-05	5.21E-09
DDE	4.83E-03	5.84E-07	9.28E-03	1.43E-06	2.23E-04	9.62E-08
DDT	9.31E-03	6.61E-08	1.33E-02	6.65E-08	1.32E-04	5.42E-09
Dibenz(a,h)anthracene	6.90E-04	9.84E-12	8.64E-04	1.23E-11	6.95E-06	2.47E-12
Di-n-butyl phthalate	1.02E-05	1.45E-09	1.43E-05	1.26E-09	5.05E-07	2.48E-10
Di-n-octyl phthalate	4.59E-06	1.31E-11	2.21E-06	1.33E-11	1.67E-08	1.57E-12
Fluoranthene	1.32E-03	1.13E-07	1.24E-03	2.66E-07	1.10E-04	3.62E-08
Fluorene	1.61E-05	8.26E-08	2.64E-05	1.55E-07	9.71E-07	2.31E-08
Indeno(1,2,3-c,d)pyrene	1.01E-02	2.39E-10	1.39E-02	3.62E-10	3.03E-05	1.15E-11
Naphthalene	2.39E-06	1.30E-08	2.25E-06	1.76E-08	1.18E-08	2.63E-09
N-Nitrosodiphenylamine	2.40E-07	3.86E-10	2.18E-07	5.86E-10	1.38E-10	2.44E-11
PCB (total)	5.00E-04	1.71E-08	7.05E-04	1.24E-08	1.48E-05	1.47E-09
Pentachlorophenol	3.61E-05	3.02E-13	4.34E-05	2.00E-13	6.34E-07	1.38E-13
Phenol	2.26E-07	1.76E-11	1.76E-07	2.29E-11	2.43E-10	2.87E-12
Pyrene	1.09E-03	1.46E-07	1.18E-03	3.87E-07	1.04E-04	4.89E-08
Antimony	1.47E-04	2.22E-20	2.06E-04	1.16E-19	6.15E-07	4.60E-21
Arsenic	1.95E-04	1.83E-22	2.90E-04	9.53E-22	8.39E-07	3.52E-22
Cadmium	7.07E-05	4.10E-22	8.07E-05	1.90E-22	7.77E-08	1.32E-23
Copper	1.48E-04	1.03E-21	2.03E-04	1.58E-21	3.49E-07	2.73E-22
Lead	2.55E-04	4.45E-24	3.55E-04	5.53E-22	7.41E-07	1.86E-22
Mercury	1.79E-03	1.64E-20	2.73E-03	2.65E-20	5.71E-06	6.93E-21
Nickel	1.50E-04	4.89E-22	2.08E-04	2.17E-21	5.26E-07	1.12E-21
Zinc	6.74E-04	7.89E-22	1.03E-03	6.36E-22	2.16E-06	1.17E-22

Table A6.2. Intake results, water, air and soil concentrations produced by Urban and Non-spatial models for emissions to air.

		1,2-Dichlorobenzene	2,4-Dimethylphenol	Acenaphthene
Non-spatial	Total Oral Intake Fraction	1.31E-07	6.80E-07	3.24E-07
	Total Inhalation Intake Fraction	4.11E-06	6.17E-08	3.14E-07
	Troposphere air concentration (kg/m^3)	3.02E-15	7.97E-16	1.53E-15
	Water concentration (kg/m^3)	2.40E-14	4.74E-14	6.47E-14
	Soil concentration in upper layers (kg/m^3)	1.41E-13	4.41E-13	4.38E-13
	Drinking water (kg/yr)	4.08E-07	8.06E-07	1.09E-06
	Unexposed produce direct (kg/yr)	1.52E-06	6.62E-05	1.50E-05
	Exposed produce direct (kg/yr)	6.75E-05	3.72E-03	9.56E-04
	Intake via eggs (kg/yr)	1.19E-09	2.44E-10	3.98E-09
	Intake via milk (kg/yr)	4.38E-09	5.35E-09	5.57E-08
	Intake via meat (kg/yr)	7.22E-08	5.91E-08	6.52E-07
Urban	Total Oral Intake Fraction	2.21E-06	9.35E-06	6.27E-06
	Total Inhalation Intake Fraction	2.71E-05	4.19E-06	1.15E-06
	Troposphere air concentration (kg/m^3)	5.28E-15	5.69E-17	2.59E-16
	Water concentration (kg/m^3)	9.68E-12	7.66E-12	2.26E-12
	Soil concentration in upper layers (kg/m^3)	2.31E-11	8.66E-12	1.36E-11
	Drinking water (kg/yr)	2.02E-04	5.09E-04	8.99E-05
	Unexposed produce direct (kg/yr)	4.42E-04	1.33E-03	6.48E-04
	Exposed produce direct (kg/yr)	1.83E-02	8.00E-02	5.33E-02
	Intake via eggs (kg/yr)	7.03E-07	6.04E-09	2.03E-07
	Intake via milk (kg/yr)	2.72E-05	1.68E-06	3.47E-05
	Intake via meat (kg/yr)	2.16E-04	2.69E-05	6.50E-04

Table A6.2. (cont'd) Intake results, water, air and soil concentrations produced by Urban and Non-spatial models for emissions to air.

		Anthracene	Benzo(a)anthracene	Benzo(a)pyrene
Non-spatial	Total Oral Intake Fraction	3.84E-06	2.50E-04	6.92E-04
	Total Inhalation Intake Fraction	2.97E-07	3.68E-07	1.70E-07
	Troposphere air concentration (kg/m^3)	1.52E-15	1.47E-15	1.23E-15
	Water concentration (kg/m^3)	1.71E-13	1.30E-12	3.89E-12
	Soil concentration in upper layers (kg/m^3)	1.71E-12	5.06E-11	4.07E-10
	Drinking water (kg/yr)	2.84E-06	1.39E-05	3.67E-05
	Unexposed produce direct (kg/yr)	7.97E-05	1.15E-04	1.11E-03
	Exposed produce direct (kg/yr)	1.20E-02	5.92E-01	2.27E+00
	Intake via eggs (kg/yr)	1.15E-07	1.22E-04	6.32E-04
	Intake via milk (kg/yr)	2.61E-06	2.95E-03	1.53E-02
	Intake via meat (kg/yr)	2.93E-05	3.33E-02	1.72E-01
Urban	Total Oral Intake Fraction	7.37E-05	4.43E-03	8.69E-03
	Total Inhalation Intake Fraction	1.06E-06	1.13E-06	6.06E-07
	Troposphere air concentration (kg/m^3)	2.34E-16	2.60E-16	1.02E-16
	Water concentration (kg/m^3)	5.84E-12	1.45E-10	9.16E-10
	Soil concentration in upper layers (kg/m^3)	5.00E-11	2.53E-09	1.47E-08
	Drinking water (kg/yr)	2.80E-04	8.84E-03	6.38E-02
	Unexposed produce direct (kg/yr)	3.12E-03	3.75E-03	1.51E-02
	Exposed produce direct (kg/yr)	6.11E-01	2.53E+01	3.76E+01
	Intake via eggs (kg/yr)	5.87E-06	2.50E-03	4.29E-03
	Intake via milk (kg/yr)	1.51E-03	6.44E-01	1.17E+00
	Intake via meat (kg/yr)	2.80E-02	1.20E+01	2.15E+01

Table A6.2. (cont'd) Intake results, water, air and soil concentrations produced by Urban and Non-spatial models for emissions to air.

		Benzo(b)fluoranthene	Butyl benzyl phthalate	Chrysene
Non-spatial	Total Oral Intake Fraction	3.50E-05	5.40E-05	2.41E-04
	Total Inhalation Intake Fraction	5.03E-08	1.52E-07	3.29E-07
	Troposphere air concentration (kg/m^3)	6.81E-16	1.08E-15	1.42E-15
	Water concentration (kg/m^3)	4.48E-13	1.59E-13	1.64E-12
	Soil concentration in upper layers (kg/m^3)	1.23E-10	5.44E-12	6.52E-11
	Drinking water (kg/yr)	5.30E-06	2.62E-06	2.15E-05
	Unexposed produce direct (kg/yr)	1.66E-05	2.25E-04	2.36E-04
	Exposed produce direct (kg/yr)	1.85E-01	2.13E-01	6.31E-01
	Intake via eggs (kg/yr)	2.84E-05	2.60E-06	6.53E-05
	Intake via milk (kg/yr)	6.85E-04	6.27E-05	1.58E-03
	Intake via meat (kg/yr)	7.72E-03	7.05E-04	1.78E-02
Urban	Total Oral Intake Fraction	5.59E-04	5.73E-04	4.24E-03
	Total Inhalation Intake Fraction	3.71E-07	6.15E-07	4.93E-06
	Troposphere air concentration (kg/m^3)	4.12E-17	1.03E-16	2.44E-16
	Water concentration (kg/m^3)	1.76E-10	2.05E-11	2.76E-10
	Soil concentration in upper layers (kg/m^3)	3.09E-09	1.22E-10	3.32E-09
	Drinking water (kg/yr)	1.57E-02	2.39E-03	2.40E-02
	Unexposed produce direct (kg/yr)	1.80E-04	4.07E-03	7.27E-03
	Exposed produce direct (kg/yr)	2.20E+00	4.67E+00	2.55E+01
	Intake via eggs (kg/yr)	2.38E-04	5.85E-05	1.75E-03
	Intake via milk (kg/yr)	7.21E-02	1.58E-02	4.53E-01
	Intake via meat (kg/yr)	9.72E-01	3.02E-01	8.59E+00

Table A6.2. (cont'd) Intake results, water, air and soil concentrations produced by Urban and Non-spatial models for emissions to air.

		DDD	DDE	DDT
Non-spatial	Total Oral Intake Fraction	5.47E-04	1.73E-04	7.53E-04
	Total Inhalation Intake Fraction	2.73E-07	8.54E-07	3.58E-07
	Troposphere air concentration (kg/m^3)	1.46E-15	2.05E-15	1.55E-15
	Water concentration (kg/m^3)	7.66E-12	1.55E-12	4.24E-12
	Soil concentration in upper layers (kg/m^3)	2.92E-10	4.11E-11	1.00E-10
	Drinking water (kg/yr)	7.36E-05	1.93E-05	3.37E-05
	Unexposed produce direct (kg/yr)	9.73E-04	3.22E-04	5.64E-04
	Exposed produce direct (kg/yr)	1.52E+00	2.62E-01	1.59E+00
	Intake via eggs (kg/yr)	4.04E-04	3.36E-05	6.27E-04
	Intake via milk (kg/yr)	9.77E-03	8.10E-04	1.52E-02
	Intake via meat (kg/yr)	1.10E-01	9.11E-03	1.71E-01
Urban	Total Oral Intake Fraction	1.30E-02	3.72E-03	1.92E-02
	Total Inhalation Intake Fraction	4.99E-06	6.67E-06	5.13E-06
	Troposphere air concentration (kg/m^3)	2.33E-16	6.41E-16	2.84E-16
	Water concentration (kg/m^3)	1.07E-09	3.15E-10	7.76E-10
	Soil concentration in upper layers (kg/m^3)	1.80E-08	3.18E-09	6.46E-09
	Drinking water (kg/yr)	6.41E-02	5.43E-03	4.29E-02
	Unexposed produce direct (kg/yr)	3.13E-02	2.09E-02	1.92E-02
	Exposed produce direct (kg/yr)	6.34E+01	2.27E+01	7.17E+01
	Intake via eggs (kg/yr)	6.83E-03	2.24E-03	8.66E-03
	Intake via milk (kg/yr)	1.79E+00	5.68E-01	2.23E+00
	Intake via meat (kg/yr)	3.43E+01	7.79E+00	4.10E+01

Table A6.2. (cont'd) Intake results, water, air and soil concentrations produced by Urban and Non-spatial models for emissions to air.

		Dibenz(a,h)anthracene	Di-n-butyl phthalate	Di-n-octyl phthalate
Non-spatial	Total Oral Intake Fraction	1.18E-04	5.10E-05	2.35E-05
	Total Inhalation Intake Fraction	1.97E-08	1.62E-07	4.91E-08
	Troposphere air concentration (kg/m^3)	4.13E-16	1.12E-15	7.11E-16
	Water concentration (kg/m^3)	3.72E-13	1.54E-13	5.39E-13
	Soil concentration in upper layers (kg/m^3)	1.84E-10	4.64E-12	7.05E-12
	Drinking water (kg/yr)	1.22E-06	2.53E-06	9.34E-08
	Unexposed produce direct (kg/yr)	5.68E-05	1.92E-04	2.69E-08
	Exposed produce direct (kg/yr)	5.54E-01	1.97E-01	6.88E-02
	Intake via eggs (kg/yr)	7.91E-04	2.64E-06	2.21E-03
	Intake via milk (kg/yr)	1.91E-02	6.38E-05	5.32E-03
	Intake via meat (kg/yr)	1.53E-01	7.17E-04	1.90E-02
Urban	Total Oral Intake Fraction	1.43E-03	5.76E-04	6.43E-05
	Total Inhalation Intake Fraction	2.16E-07	6.39E-07	3.25E-07
	Troposphere air concentration (kg/m^3)	1.71E-17	1.10E-16	3.25E-17
	Water concentration (kg/m^3)	1.91E-10	1.62E-11	2.18E-10
	Soil concentration in upper layers (kg/m^3)	3.56E-09	1.03E-10	1.56E-10
	Drinking water (kg/yr)	4.74E-03	1.84E-03	2.85E-04
	Unexposed produce direct (kg/yr)	3.80E-04	3.72E-03	1.86E-07
	Exposed produce direct (kg/yr)	3.68E+00	4.68E+00	5.02E-01
	Intake via eggs (kg/yr)	6.15E-04	6.31E-05	9.87E-05
	Intake via milk (kg/yr)	2.07E-01	1.70E-02	3.15E-03
	Intake via meat (kg/yr)	1.71E+00	3.30E-01	3.44E-02

Table A6.2. (cont'd) Intake results, water, air and soil concentrations produced by Urban and Non-spatial models for emissions to air.

		Fluoranthene	Fluorene	Indeno(1,2,3-c,d)pyrene
Non-spatial	Total Oral Intake Fraction	1.92E-04	1.05E-06	1.25E-05
	Total Inhalation Intake Fraction	6.38E-07	3.06E-07	1.48E-08
	Troposphere air concentration (kg/m^3)	1.94E-15	1.52E-15	3.42E-16
	Water concentration (kg/m^3)	1.55E-12	1.12E-13	4.13E-12
	Soil concentration in upper layers (kg/m^3)	3.52E-11	8.33E-13	1.39E-10
	Drinking water (kg/yr)	2.36E-05	1.89E-06	1.48E-05
	Unexposed produce direct (kg/yr)	1.07E-03	3.68E-05	2.69E-07
	Exposed produce direct (kg/yr)	3.95E-01	3.20E-03	3.48E-02
	Intake via eggs (kg/yr)	1.68E-05	1.60E-08	4.47E-05
	Intake via milk (kg/yr)	4.05E-04	3.13E-07	1.08E-03
	Intake via meat (kg/yr)	4.54E-03	3.53E-06	9.64E-03
Urban	Total Oral Intake Fraction	4.16E-03	2.01E-05	8.81E-03
	Total Inhalation Intake Fraction	2.16E-06	1.10E-06	3.20E-06
	Troposphere air concentration (kg/m^3)	5.14E-16	2.46E-16	1.35E-17
	Water concentration (kg/m^3)	1.53E-10	3.81E-12	1.17E-09
	Soil concentration in upper layers (kg/m^3)	2.11E-09	2.50E-11	3.25E-09
	Drinking water (kg/yr)	7.68E-03	1.63E-04	3.76E-02
	Unexposed produce direct (kg/yr)	6.21E-02	1.51E-03	1.51E-06
	Exposed produce direct (kg/yr)	3.09E+01	1.71E-01	1.91E-01
	Intake via eggs (kg/yr)	1.32E-03	8.43E-07	4.51E-05
	Intake via milk (kg/yr)	3.35E-01	1.92E-04	1.74E-02
	Intake via meat (kg/yr)	5.06E+00	3.55E-03	1.04E-01

Table A6.2. (cont'd) Intake results, water, air and soil concentrations produced by Urban and Non-spatial models for emissions to air.

		Naphthalene	N-Nitrosodiphenylamine	PCB (total)
Non-spatial	Total Oral Intake Fraction	1.26E-08	1.25E-09	8.82E-04
	Total Inhalation Intake Fraction	7.00E-08	1.39E-08	7.04E-07
	Troposphere air concentration (kg/m^3)	8.31E-16	3.25E-16	1.66E-15
	Water concentration (kg/m^3)	6.57E-15	1.57E-16	8.06E-13
	Soil concentration in upper layers (kg/m^3)	4.59E-14	9.35E-15	9.71E-12
	Drinking water (kg/yr)	1.12E-07	2.67E-09	4.90E-07
	Unexposed produce direct (kg/yr)	1.27E-06	2.13E-07	2.89E-06
	Exposed produce direct (kg/yr)	6.62E-05	9.00E-06	6.06E-01
	Intake via eggs (kg/yr)	3.34E-10	6.93E-11	5.47E-03
	Intake via milk (kg/yr)	1.83E-09	2.68E-10	4.69E-02
	Intake via meat (kg/yr)	2.63E-08	4.33E-09	1.67E-01
Urban	Total Oral Intake Fraction	2.00E-07	1.38E-08	1.60E-03
	Total Inhalation Intake Fraction	4.31E-06	3.49E-06	1.47E-06
	Troposphere air concentration (kg/m^3)	6.62E-17	1.52E-17	3.80E-16
	Water concentration (kg/m^3)	1.31E-13	5.16E-15	2.30E-11
	Soil concentration in upper layers (kg/m^3)	8.02E-13	1.23E-13	9.07E-11
	Drinking water (kg/yr)	9.81E-06	2.65E-07	2.46E-04
	Unexposed produce direct (kg/yr)	2.89E-05	2.83E-06	2.38E-05
	Exposed produce direct (kg/yr)	1.70E-03	1.18E-04	8.86E+00
	Intake via eggs (kg/yr)	3.83E-09	2.60E-10	7.99E-04
	Intake via milk (kg/yr)	4.23E-07	2.62E-08	2.08E-01
	Intake via meat (kg/yr)	7.90E-06	2.58E-07	3.29E+00

Table A6.2. (cont'd) Intake results, water, air and soil concentrations produced by Urban and Non-spatial models for emissions to air.

		Pentachlorophenol	Pyrene	Mercury
Non-spatial	Total Oral Intake Fraction	1.75E-04	1.43E-04	1.88E-04
	Total Inhalation Intake Fraction	1.51E-07	6.89E-07	7.87E-06
	Troposphere air concentration (kg/m^3)	1.15E-15	1.98E-15	3.95E-15
	Water concentration (kg/m^3)	2.16E-12	1.35E-12	1.56E-11
	Soil concentration in upper layers (kg/m^3)	5.80E-11	2.67E-11	2.00E-08
	Drinking water (kg/yr)	3.67E-05	2.07E-05	1.64E-04
	Unexposed produce direct (kg/yr)	1.52E-02	8.32E-04	7.86E-05
	Exposed produce direct (kg/yr)	6.59E-01	2.83E-01	1.48E-04
	Intake via eggs (kg/yr)	9.69E-08	1.10E-05	3.27E-09
	Intake via milk (kg/yr)	2.35E-06	2.64E-04	6.04E-07
	Intake via meat (kg/yr)	2.55E-05	2.96E-03	8.98E-06
Urban	Total Oral Intake Fraction	9.16E-04	3.31E-03	4.45E-04
	Total Inhalation Intake Fraction	3.93E-06	6.50E-06	9.54E-06
	Troposphere air concentration (kg/m^3)	6.82E-17	5.95E-16	1.33E-15
	Water concentration (kg/m^3)	7.29E-10	1.55E-10	6.91E-09
	Soil concentration in upper layers (kg/m^3)	1.17E-09	1.77E-09	9.12E-06
	Drinking water (kg/yr)	8.80E-02	7.63E-03	1.15E-01
	Unexposed produce direct (kg/yr)	1.46E-01	5.41E-02	4.77E-03
	Exposed produce direct (kg/yr)	6.99E+00	2.49E+01	1.03E-02
	Intake via eggs (kg/yr)	1.27E-06	1.02E-03	2.85E-07
	Intake via milk (kg/yr)	3.36E-04	2.57E-01	8.83E-04
	Intake via meat (kg/yr)	5.62E-03	3.75E+00	1.26E-03