A literature review of risks relevant to the use of biosolids and compost from biosolids with relevance to the Nicola Valley, BC

Contract # CS16JHQ112

Proposal Prepared for:

British Columbia Ministry of Environment

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March 2016

Executive summary

This report reviews the state-of-science in the assessment of the risks associated with the land application of biosolids and its derivatives in the context of the Nicola Valley, BC. It is our intent to highlight knowledge gaps and identify best practices for reducing the risk associated with land application of biosolids and its derivatives. A full risk assessment of the land application of biosolids in the Nicola Valley was beyond the scope of this work.

The biophysical and social conditions in Nicola Valley provide an important context to frame the use of biosolids in this region. Knowledge of climatic, soils and their distribution, hydrology, people and land use in the Nicola Valley were summarized to provide an understanding of the context. In particular, it was noted that while the climate of the Nicola Valley is generally arid reducing the risk of water contamination, there is significant seasonal groundwater recharge and the application of biosolids should reflect this seasonality. Further, the majority of land uses to which biosolids are being applied are for the production of hay or grazing of animals.

A solid foundation of research and policy on the management of biosolids exists in Canada and has culminated in a Canada-wide approach for the management of wastewater biosolids. This framework encourages the beneficial use and sound management of biosolids. In our review we have focused on the risks not addressed within this framework such as the issue of emerging substances of concern (ESOC). Further we have highlighted best practices in treatment and land application guidelines to further reduce the risks associated with the land application of biosolids.

The toxic materials found in biosolids are there because we put them there. Source reduction initiatives remain a best practice for reducing the toxicity of human sewage and the biosolids that are produced from it. It is important to note that the concentrations of some of the ESOCs that are found in composted biosolids are lower than the maximum amounts allowed in personal care products such as soap and tooth paste. Another important source of active pharmaceutical ingredients found in biosolids are linked to improper disposal of these products. Programs to reduce or eliminate these disposal pathways are also a best practice.

Monitoring and reporting of the composition of biosolids and composted biosolids, particularly the presence of ESOCs, are important tools for managing and communicating the potential for environmental risk. It is critical that results of such a testing program should be made available to the public to provide oversight and to address public concerns as to the composition of the material being applied to land.

Managing exposure pathways is an important risk management tool. In the context of the application of biosolids in the Nicola Valley, reduced exposure of workers and adjacent residents to bioaerosols and odours was seen as a best practice. Another important exposure pathway was the direct injection of biosolids or plant covered biosolids by grazing animals. Direct injection or immediate soil incorporation of applied biosolids was seen as a best practice for reducing this exposure pathway. Seasonal patterns in precipitation influence the potential for biosolids impact on aquatic ecosystems. The greatest risk for ground and surface water contamination occurs immediately after land application. A best practice is to avoid land-application of biosolids when the groundwater table is high during late fall, winter and spring, and when there are concerns about potential runoff.

Currently, composting is one of the primary biosolids treatment processes occurring in the Nicola Valley. The literature demonstrates that composting can effectively reduce pathogenic bacteria, viruses and parasites in biosolids. Although pathogens cannot be eliminated completely by the composting process, levels are frequently lower than those found in raw livestock manures, which are applied annually. While the concentrations of heavy metals are increased through the composting process, their availability is reduced and thus their potential for environmental impact. While research is still limited, several of the organic compounds identified as ESOCs can be effectively reduced by the composting process. There is a need for additional research to document that the primary ESOCs present in the biosolids being applied in the Nicola Valley are reduced or eliminated as a result of the compositing process.

In examining the literature on the thermal treatment of biosolids to reduce environmental risks, it was noted that this research is promising but still inconclusive. There are a number of processes that have been demonstrated to reduce environmental risks associated with metals, pathogens and ESOCs, but in some cases thermal treatment can result in the production of toxic organics. Further, an engineering assessment of the feasibility of thermal treatment should be undertaken before these technologies can be recommended.

Ecosystems are complex and risk depends on a multitude of predictable and unpredictable factors. Moving forward, it is important to consider that natural and artificial chemicals are everywhere in the environment around us. Humans are exposed to high levels of cadmium in jewellery and potent anti-microbial agents in hygiene products. Simply detecting a particular contaminant does not provide an accurate representation of its ecological and human health risk. Comparing the risk of biosolids to other point and nonpoint sources of contaminant exposure, such as animal manures, is critical in evaluating the potential use of biosolids as soil amendment and their role in efficiently recycling human waste. Biosolids-derived contaminants can pose a risk for wildlife and aquatic life. More research is needed to understand, first, the potential for land-applied biosolids to contaminate ground and surface-water, and secondly, the potential impact on wildlife and natural aquatic ecosystems.

Knowledge Gaps

1. There is a lack of comprehensive empirical datasets for detailed risk assessment, and current approaches frequently depend on safety factors rather than a substantial body of directly relevant data based on empirical observation.
2. There is very limited field-based research available that looks at the potential containment exposure to aquatic ecosystems following land application of biosolids in nearby grasslands or pastures. More research is needed to understand, first, the potential for land-applied biosolids to contaminate ground and surface-water in the Nicola Valley, and secondly, the potential impact on natural aquatic ecosystems with wild populations.
3. More comprehensive research is required to validate and understand the relative risk of biosolids compared to other organic wastes commonly applied to agricultural land across the province
4. There are limited comprehensive ecological studies that examine the effect of biosolids-application on natural land and aquatic ecosystems, including wildlife at all trophic levels.
5. There is a gap in the literature pertaining to the synergetic effect of mixtures of organic and metal contaminants and risk to humans and the environment (Arnold et al. 2013; Brooks et al. 2012a; Backhaus and Karlsson 2014; Roccaro et al. 2014).
6. There is a lack of controlled studies which compare the raw or treated biosolids to a composted product and describe the impacts of these products on different soil systems (Briceno et al. 2007). An additional challenge is the ever expanding list of 'emerging' contaminants that could, and should, be tested.
7. While thermal treatment may prove to be a successful option for the safe utilization of biosolids, the reliability and practicality of those approaches to processing the biosolids being land applied in the Nicola Valley has yet to be demonstrated. It is critical that the ability of these treatment processes to reduce toxic loading through out the the life cycle (air, water, soil and biota) be clearly documented before they can be considered as viable alternatives.

Recommendations

1. There should be routine reporting of the amount and composition of biosolids and composted biosolids, including priority ESOCs, being applied to land in the Nicola Valley. This information should be available to the public.
2. Source reduction initiatives should continue to be a primary focus in reducing the toxicity of materials entering the waste stream and the need for the land disposal of these compounds.
3. That a quantitative risk assessment of land application of biosolids specific to the Nicola Valley be conducted. This risk assessment should consider pathways H1A, H6B, H7, E1B, E4 that either of concern based on studies reported in the literature or are poorly represented in the literature and are of particular concern to residents in the Nicola Valley. The risk assessment should include consideration of risks associated with exposure to candidate pharmaceuticals and personal care products (Table 13).
4. An organic contaminant monitoring program should be established for biosolids being applied in the Nicola Valley. This should include monitoring of concentrations in source materials and treated products.
5. Should direct ingestion by domestic and wildlife be found to be an exposure pathway of concern, the feasibility of the incorporation of land applied biosolids animals should be examined as a means of reducing direct exposure of biosolids to grazing.
6. Consideration should be given to extending the OMRR regulations pertaining to set back distances to water courses for the land application of Class B biosolids and Class B composts to other products controlled under the OMRR.
7. The fate of major emerging substances of concern during composting should be documented.

1. *Preface*

This report was compiled by Land Resource Consulting Services for the British Columbia Ministry of the Environment to review the state-of-science in support of an assessment of the risk associated with the land application of biosolids in the Nicola Valley, British Columbia. In response to the concerns raised by First Nations and stakeholders regarding the storage and use of biosolids and compost from biosolids, the Ministry of Environment undertook a scientific review of the land application of biosolids in the Nicola Valley. The Province committed to considering regulatory, policy or practice changes based on the findings of the review. A Technical Working Group (TWG) has been formed to conduct the scientific review of the use of biosolids including land application of biosolids and compost containing biosolids.

The object of the review was to "produce a product that encapsulates a sufficient body of information that the scientific experts, governments, First Nations and the public can review and to inform their opinions on the practice of land applying biosolids and compost from biosolids". As part of that review we were asked to identify knowledge gaps and highlight best practices to reduce risk.

Specifically, were asked to:

- Review of the existing research (might consist of the review of the Literature Reviews and some individual research) related to how biosolids and compost from biosolids may impact wildlife (the Literature Review and research on the effect of biosolids and compost from biosolids on domestic animals might also help explain the effects of biosolids and compost from biosolids on wildlife);

- Explore the possibility for the wildlife to be a pathway for potential contaminants of biosolids and compost from biosolids to enter the human food chain and affect human health.

- Perform a scientific review of the research and risk assessments regarding the benefits and quantitative human health risks associated with land applying biosolids and compost from biosolids; and

- Perform a scientific review of any alternative management methods regarding the management of biosolids and compost from biosolids.

This report does not provide a risk assessment of the use biosolids in the Nicola Valley. A risk assessment specific to the Nicola Valley requires more detailed information on the quantity, composition and treatment of the biosolids being land applied; the land uses and land application methods being used; and a detailed characterization of the "at risk" populations. This report provides a review of the current state-of-science that would inform such a risk assessment.

"Everything that we use comes out of the earth and whatever we are finished with goes back into it."

David Suzuki, The Bottom Line, CBC Radio, March 10 2011

Appropriate management for human sewage is not finding some deep, dark hole where we can "dispose" of this material "out of sight, out of mind" but rather safely harvesting the energy and returning the nutrients contained in this resource to the ecosystem. The issue is not ***whether*** the nutrients contained in human sewage should be returned to land, but ***how*** and ***where*** it can be done safely and sustainably. The recycling of human sewage sludge to land to gain benefit from the essential plant nutrients and organic matter it contains, is regarded by most scientific and regulatory authorities as the most pragmatic and environmentally sustainable approach to managing the sludge generated from urban wastewater treatment (Smith 2009; Clarke and Smith 2011; Clarke and Cummins 2015). The components that are of concern in human sewage, or the biosolids that are produced from it, are added to the waste stream as a result of human choices and can be removed from it, to a greater or lesser extent, by appropriate source reduction initiatives, handling or treatment (Joo et al. 2015).

Much like "reduce" is the first of the three Rs of recycling, we should not ignore the opportunities to reduce toxic components at source. Additives in personal care products, flame retardants added to our clothing, pharmaceuticals in our medications and metals from industrial processes are introduced to our waste stream as a result of human choices. It is only when they appear in sewage that we become concerned about their potential for environmental impact, often at concentrations much lower than occurred in the original regulated use. If it is of concern when it is present in our soil, why is not also of concern when it appears in our toothpaste, on our clothing and in our medications? Our greatest opportunity to solve this problem occurs before we push the level to flush the problem "out of sight, out of mind". Initiatives such as British Columbia's Medications Return Program are attempting this source reduction strategy.

In this review we have taken the approach that fundamentally the nutrients contained in human sewage should be returned to the land in the form of treated biosolids (ideally the land from which they came). We have reviewed the scientific literature as to how or whether this can be done safely given the current composition of the waste stream. We reviewed the various processing technologies that modify raw sewage in an attempt to reduce the risk of human and environmental impact of this material. Ultimately how we utilize (or dispose) of human sewage is a political decision. We hope that the information presented in this review is useful in informing those decisions so as to ensure that result is safe and sustainable.

2. *The Context - An overview of Nicola Valley ecosystem and the use of biosolids in the Nicola Valley*

The Nicola Valley is located in the Southern Interior of British Columbia in Thompson Country and is often referred to as the Thompson-Nicola Regional District (Fig. 1).

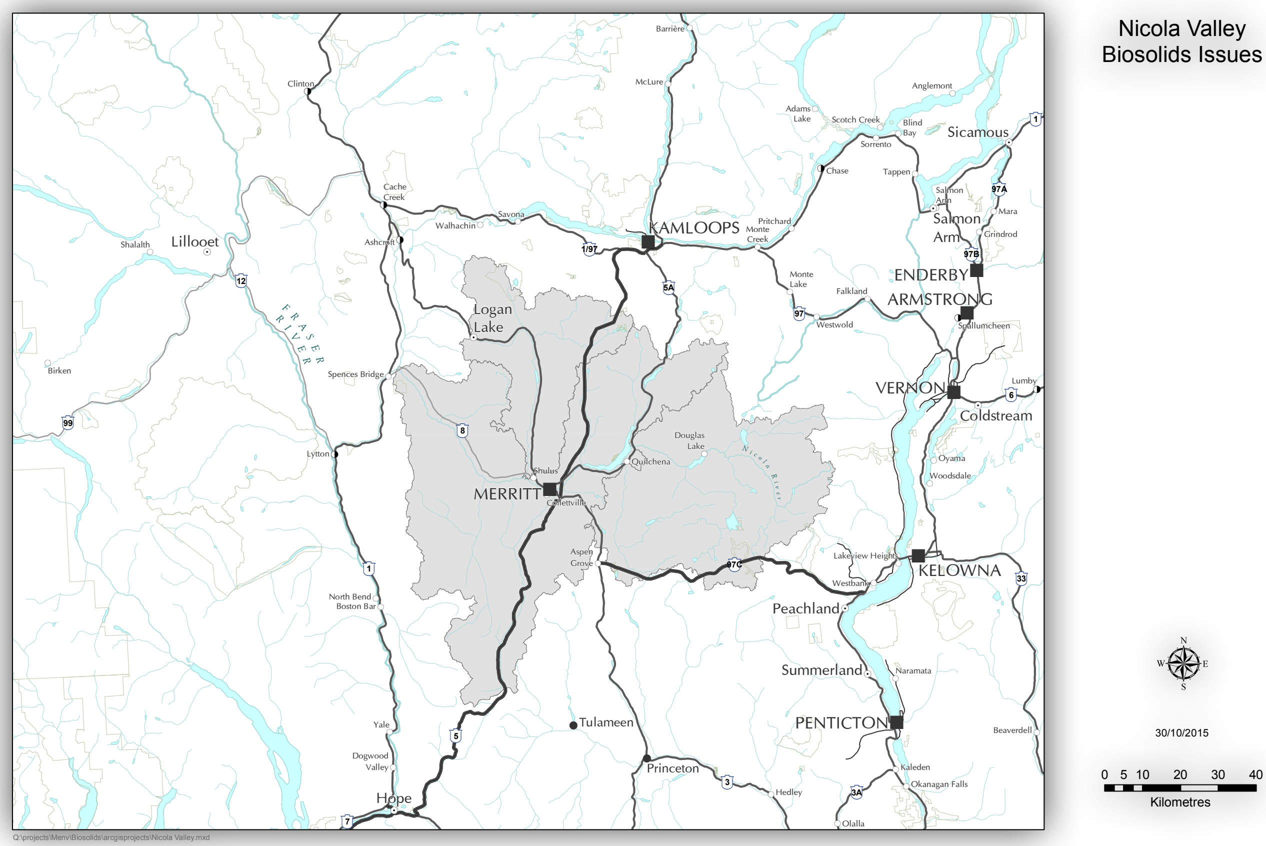

Figure 1: The Nicola Valley area of interest in terms of biosolids application (Source: BCME).

The valley is located in the rain shadow of the Canadian Cascades and the Lillooet Ranges of the Coast Mountains. The climate is dry (30-year normal are 254 mm rainfall; 66.7 mm snowfall; 321 mm precipitation) and warm (average temperature 7.8 ºC) climate (Fig. 2). Precipitation is somewhat seasonal with greatest amounts occurring in the spring and fall (Fig. 2).

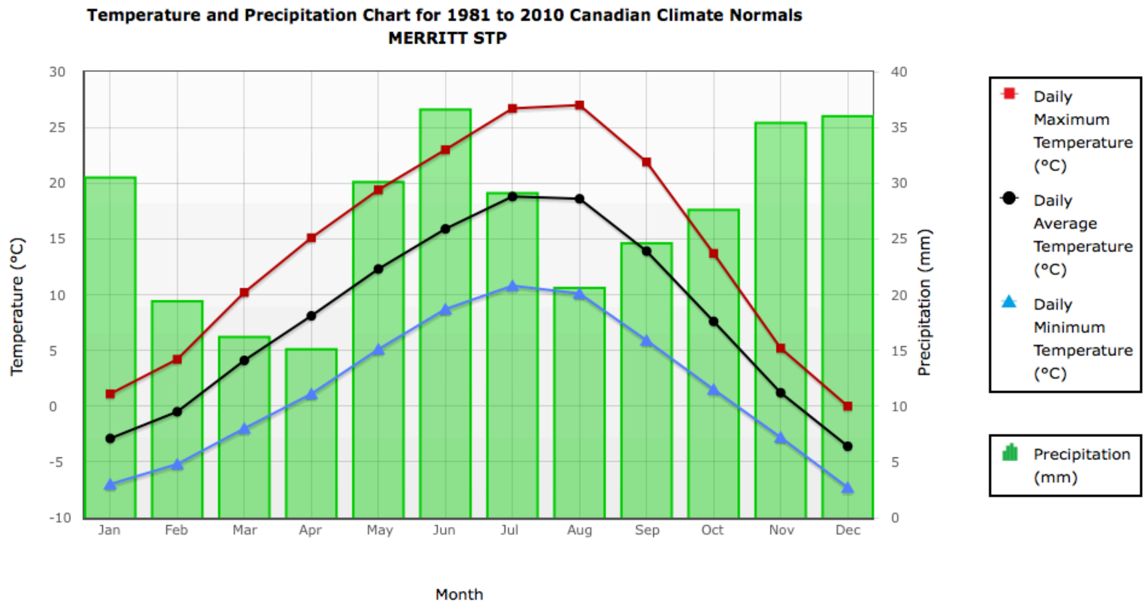

Figure 2: Monthly 30-year normal temperature and precipitation for Merritt, BC located in the Nicola Valley (Environment Canada).

The Nicola Valley is part of the Nicola Basin Ecosection (NIB), located with in the Southern Interior Ecoprovince (SOI) of the Thompson – Okanagan Pleateau (TOP) Ecoregion. Because of its lower elevation (627 m at Nicola Lake) and rain shadow effects this area is dominated by sagebrush-steppe, bunchgrass-steppe and meadow-steppe communities, with dry ponderosa pine stands on the adjacent slopes. Douglas-fir grows as single stands in the moister draws and gullies and as dense stands on north-facing slopes and higher elevations. All the pine forests have been heavily impacted by the mountain pine beetle. In the very dry valley bottoms of the southern Okanagan, antelope-brush and prickly-pear cactus dominate the lower sites. (BCME 2016a)

Soils – Grassland soils are dominant, having developed on sites varying from coarse gravel to silt. Those soils are often calcareous, with dark brown to black surface layers, and are rich in organic matter (BCME 2016b). The soils of the region include areas that are dominantly by Brunisolic (Eutric), Luvisolic (Gray), and Chernozemic (Dark Brown) soils (Figs. 3 & 4).

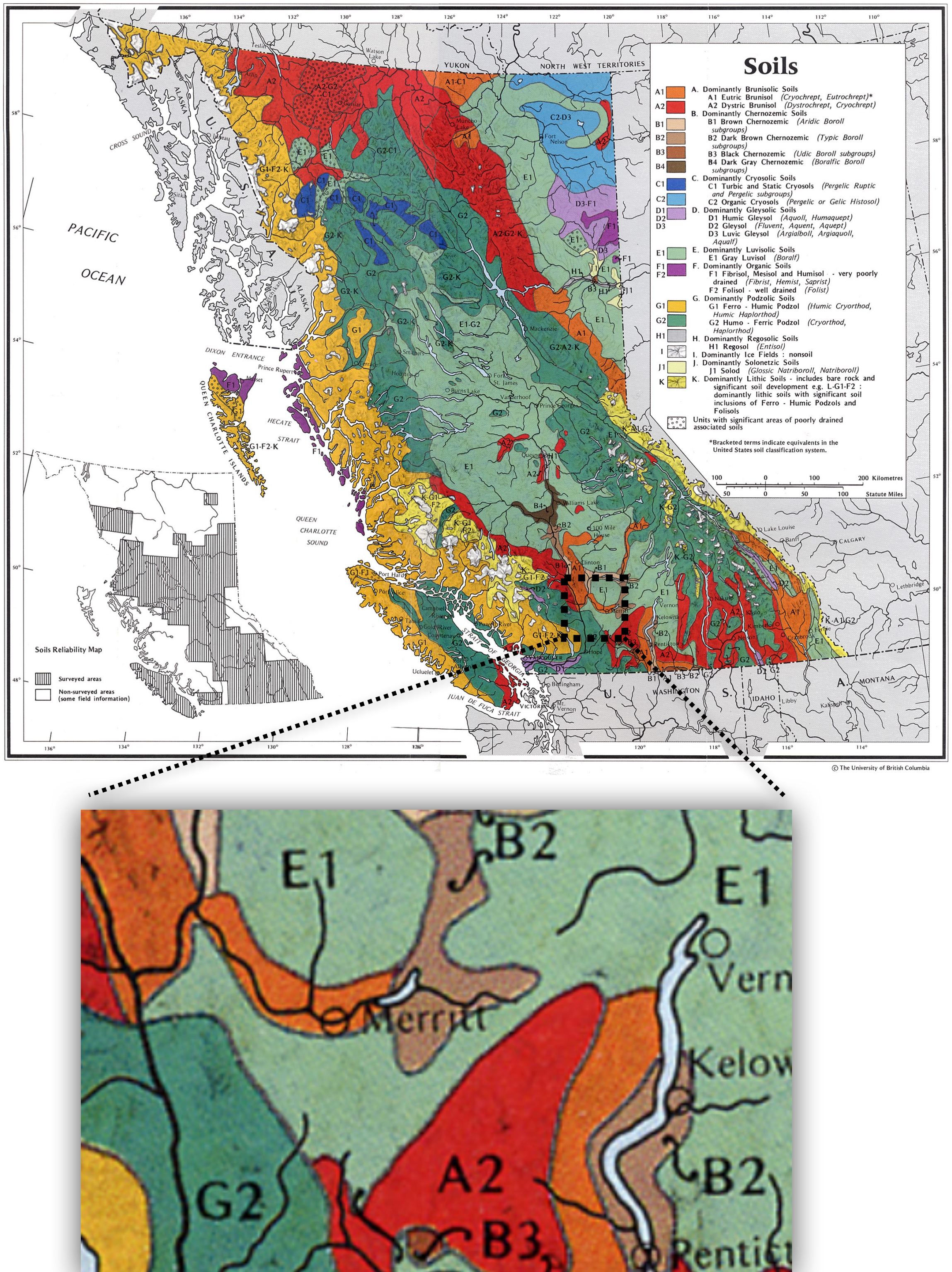

Figure 3: Map of the distribution of soil in the Nicola Valley area. Orange represents areas dominated by Eutric Brunisols (orange), Gray Luvisols (green), and Dark Black Chernozemic (brown). ([BCSIS 2016]())

Figure 4: Soil Landscapes on the Southern Plateau Near Merritt (BCME 2016b). Numbers indicate the distribution of soils in the landscape as described below.

Humic Regosol and Humic Gleysol landscapes on the present river floodplain, with medium textured parent materials. The vegetation consists of sedges, and willows.

1. Dark Brown soil landscape on lacustrine silts with grassland vegetation. There are some pockets of the Solodized Solonetz soil landscape with salt tolerant vegetation.
2. Dark Brown soil landscape on gravelly medium textured glacial grassland vegetation.
3. Black soil landscape on gravelly medium textured glacial till. The vegetation is mainly grassland with tree islands of trembling aspen and Rocky Mountain Douglas-fir.
4. Eutric Brunisol landscape on gravelly medium textured glacial till and colluvium with an open Rocky Mountain Douglas-fir - pinegrass vegetation cover.
5. Gray Luvisol landscape on gravelly medium textured glacial till. There is a relatively dense Rocky Mountain Douglas-fir - pinegrass vegetation cover.

Hydrology – The Nicola Basin is serviced by a surficial (unconfined) aquifer of approximately 10 km^2 and services a population of 3,000 – 10,000, mainly the City of Merritt (BCME). The rivers are of high fishery value. Water access during the arid summer period has become an issue. Peak flows in the region occur from April to June (BCME 1988). In 1992 in the Merritt area, roughly half of the wells are completed in bedrock and half are completed in unconsolidated deposits (Environment Canada 1992).

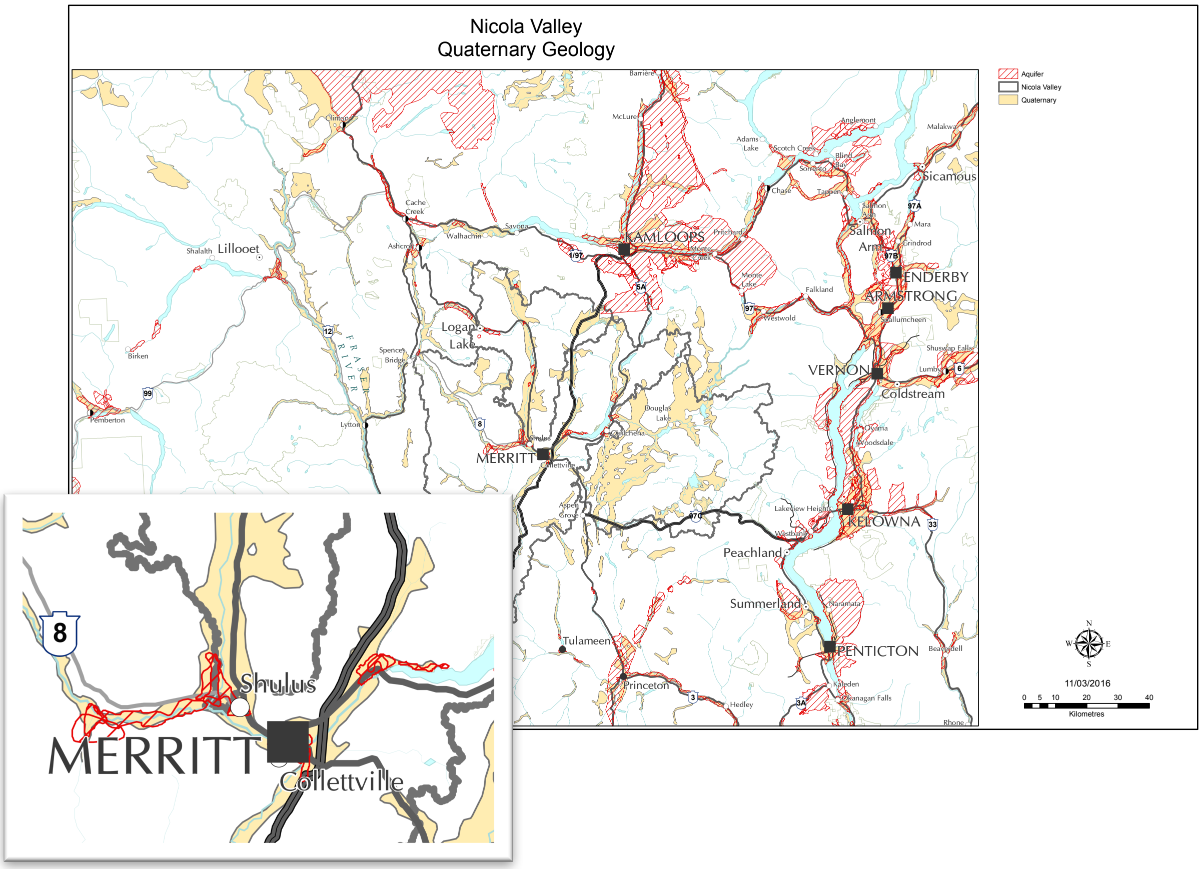

Figure 5: Quaternary geology and aquifer locations in the Nicola Valley Region (source BCME 2016)

Land use – The dominant agricultural land use is rangeland and hay production.

The climate and soils of the region provide important context for the assessment of the risk associated with land application of biosolids. Firstly, the arid climate with seasonal periods of runoff and groundwater recharge suggest that the potential for impact upon on both groundwater and surface water will be very seasonal and the timing of application should be made to reflect this. Secondly, the high organic matter content of the soils will likely translate into a higher than average ability for degradation and sorption of the metals and organic contaminants contained in biosolids following land application. Thirdly, the coarse texture of the subsurface deposit suggests a limited capacity to attenuate compounds that migrate beyond the surface soil to impact groundwater. Finally, the application of biosolids to rangeland and land used for the production of hay mitigates the potential for surface run off but highlights the importance of direct ingestion by domestic and wild animals.

People – The Nicola Valley is home to five First Nations (Nooaitch, Lower Nicola, Shackan, Coldwater, and Upper Nicola Indian Bands). There are several large ranches in the area including the Douglas Ranch (aka Douglas Lake Cattle Company). There are also residential subdivisions within the area. The City of Merritt is home to approximately 7,000 residents.

The Use of Biosolids in the Nicola Valley

The use of biosolids in the Nicola Valley has become controversial. In 2014, local residents and First Nations have objected to the processing and land application of biosolids. In 2015 the BC Government announced it would conduct a scientific review of the use of biosolids in the Nicola Valley. This review of the scientific literature is part of that review.

According to information available at the time of writing, in 2013 and 2014, the total application of biosolids in the Nicola Valley included 31,000 tonnes of material that was incorporated into the remediation of mine sites and over 37,000 tonnes that were land applied in agricultural land uses (Table 1). There does not appear to be a structured process for the recording and report of the application of biosolids and their composition.

Table 1: Land application of biosolids in the Nicola Valley in 2013 and 2014 (BCME pers. comm.)

Source	Treatment	Land Application	2013	2014
			(dry tonnes)	
Metro Vancouver	Digested	Surface application to Pasture and Rangeland	3,636	15,028
	Digested	Incorporated on reclaimed mine tailings	24,480	7,012
Central Okanagan	Composted	Surface application to Pasture and Rangeland	42*	1,350
Chilliwack	Digested	Surface application to Pasture and Rangeland	N/A	2,360

* 42 bulk tonne is only a portion of the biosolids that was delivered. A land application was submitted by Central Okanagan and some amount of biosolids was deliver in 2013. No access to data at this point.

Metro Vancouver provided information on the amounts and treatment of the biosolids being applied in the Nicola Valley (both agricultural and reclamation) indicating that much of the material produced qualified as a Class A biosolids which is not regulated under the BC Organic Matter Recycling Regulation (Table 2; BC 2002). The stabilization method was aerobic or anaerobic digestion.

Table 2: Biosolids application in the Nicola Valley 2013 and 2014 (Metro Vancouver pers. comm.)

Metro Vancouver Biosolids - Applied in the Nicola Valley 2013 2014

Wastewater Treatment Plant	**Annacis Island**	**Lulu Island**	**Lions Gate**	**Northwest Langley**	**Iona Islan**
Biosolids Class	A	B	B	B	B
Treatment Level	secondary	secondary	primary	secondary	primary
Stabilization Method	Thermophilic anaerobic digestion	Mesophilic anaerobic digestion	Thermophilic anaerobic digestion	Mesophilic aerobic digestion	Mesophil anaerobi digestior
Avg. time, temp.	28 days, 55C	33 days, 37C	17 days, 55C	40 days, 42C	17 days, 3
Bulk Tonnes Applied in Nicola Valley					
2013 surface applied	2795	841	0	0	0
2013 incorporated	13686	3680	961	5598	1555
2013 Total	***16481***	***4521***	***961***	***5598***	***1555***
2014 surface applied	13251	1580	197	0	0
2014 incorporated	6103	909	0	0	0
2014 Total	***19354***	***2489***	***197***	***0***	***0***

There is currently not a central repository to record the amounts or composition of biosolids being applied to land in British Columbia. We were unable to locate any routine reporting of the composition of biosolids including metal and organic contaminant concentrations for biosolids being applied in the Nicola Valley. While these records likely exist, they were not publically available at the time of writing.

Recommendation 1: There should be routine reporting of the amount and composition of biosolids and composted biosolids, including priority ESOCs, being applied to land in the Nicola Valley. This information should be available to the public.

3. *Regulatory approaches and supporting materials for the management of wastewater biosolids in Canada, the United States and European Union.*

Determining the risk associated with a practice and its sustainability is limited by available information and the state-of-science. In the case of land application of biosolids there is a rich foundation of information on nutrient, metal and pathogen impacts, both in terms of reports commissioned by various regulatory agencies and reviews in the scientific literature, however the occurrence and risk associated with organic contaminants is still an emerging area of research. There is still inadequate understanding of risk associated with a wide range of organic compounds and more research is required.

Also the risk associated with not traditionally considered exposure pathways (e.g., wildlife exposure) is limited.

The following information resources document the regulatory approaches that have been adopted in various jurisdictions based on current science.

Canadian Council of Ministers of the Environment

The Canadian Council of Ministers of the Environment (CCME) is an inter-governmental forum for action on environmental issues of national and international concern. In 2008, the CCME undertook the development of a Canada-wide approach for the management of municipal wastewater biosolids. To inform their work, the CCME conducted reviews of: the scientific literature (CCME 2012a); a jurisdictional review of the legislative framework in Canada (CCME 2010); baseline data on the composition of biosolids in Canada (Hydromantis 2009); and a Canada-wide approach for the land application of biosolids (CCME 2012b).

As part of their work, the CCME developed policy and principals for beneficial use of biosolids and invited feedback on this work through two phases of public consultation in 2010 and 2011. Subsequently, the Ministers approved the Canada-wide Approach for the Management of Wastewater Biosolids in 2012. The Approach encourages the beneficial use and sound management of biosolids. This information provides a firm knowledge base to inform science-based decisions relating to wastewater management and allows for the implantation of uniform approaches to beneficial uses of biosolids in Canada. The adoption of these guidelines occurs at the Provincial and Territorial level. Note that the Canada-wide approach merely provides a framework for biosolids management, the adoption, delivery and enforcement of specific regulations for biosolids management is a Provincial and Territorial government jurisdiction.

The following documents provide a knowledge resource for wastewater biosolids management as well as outline a proposed Canada-wide approach to the management of wastewater biosolids.

- Canada-wide Approach for the Management of Wastewater Biosolids (2012) CCME (2012a)
- Guidance Document for the Beneficial Use of Municipal Biosolids, Municipal Sludge and Treated Septage (2012) CCME (2012b)
- A Review of the Current Canadian Legislative Framework for Wastewater Biosolids (2010) CCME(2010)
- Emerging Substances of Concern in Biosolids: Concentrations and Effects of Treatment Processes (2009) Hydromantics (2009)
- Emerging Substances of Concern in Biosolids: Concentrations and Effects of Treatment Processes - Field Sampling Program Final Report (2010) Hydromantics (2010)

- The Biosolids Emissions Assessment Model (BEAM): A Method for Determining Greenhouse Gas Emissions from Canadian Biosolids Management Practices (2009) SYLVIS (2009)
    - Biosolids Emissions Assessment Model (BEAM) 1.1 (2011)

Regulation of Biosolids in British Columbia
In British Columbia biosolids are regulated under the Organic Matter Recycling Regulation (OMRR; BC 2002), developed in 2002 under the authority of the Environmental Management Act and the Health Act.

The following activities are regulated under the OMRR:
- the production, distribution, sale, storage, use and land application of biosolids and compost; and,
- the construction and operation of composting facilities. Approvals or permits from the BC Ministry of Environment are required for organic materials and processes not covered by the OMRR, for biosolids that do not meet the pathogen or trace element limits required by the OMRR, and for the application of biosolids or compost to soils that contain elevated trace element concentrations exceeding the standards specified in the OMRR.

The OMRR is designed to ensure that residuals are used in a manner protective of human health and the environment. The most appropriate use of residuals was considered to be as a nutrient source or soil conditioner, as opposed to disposal as a waste material.

The OMRR defines biosolids as stabilized municipal sewage resulting from municipal wastewater or septage treatment that has been sufficiently treated to reduce pathogen densities, vector attraction and trace element concentrations. As such, the term biosolids in BC is restricted to certain municipal wastewater-based products. Biosolids must meet the following scheduled requirements stipulated in the OMRR:
- pathogen reduction processes outlined in Schedule 1;
- vector attraction reduction processes outlined in Schedule 2;
- pathogen reduction limits specified in Schedule 3;
- quality criteria outlined in Schedule 4;
- sampling and analysis protocols and frequencies provided in Schedule 5; and,
- record keeping practices specified in Schedule 6.

The OMRR provides for the beneficial use of biosolids that meet these specific quality criteria. Biosolids are classified in the OMRR as either Class A or Class B biosolids, depending on the extent to which these process and quality-based criteria are met (Table 3). The class of biosolids achieved has implications for recycling options, final land use, site access, application methodology and monitoring requirements following biosolids application.

Table 3: Composition criteria for class A and class B biosolids in British Columbia as defined by the Organic Matter Recycling Regulation (BC 2002).

OMRR Criteria	Class A Biosolids	Class B Biosolids
Process Criteria (Schedules 1, 2 and 3)		
Pathogen Reduction	√	√
Vector Attraction Reduction	√	√
Fecal Coliform (MPN g^{-1} dw)	< 1,000	< 2,000,000
Quality Criteria (Schedule 4; $\mu g\ g^{-1}$, dw)[a]		
Arsenic	75	75
Cadmium	20	20
Chromium	-	1,060
Cobalt	150	150
Copper	-	2,200
Lead	500	500
Mercury	5	15
Molybdenum	20	20
Nickel	180	180
Selenium	14	14
Zinc	1,850	1,850
Foreign Matter	≤ 1%[b]	≤ 1%[b]
Sampling and Analyses and Record Keeping (Schedules 5 and 6)		
Sampling and Analysis	Required	Required
Record Keeping	Required	Required

[a] Class A Limits for trace elements specified in *Trade Memorandum T-4-93 (September 1997), Standards for Metals in Fertilizers and Supplements*.

[b] Further requirement of no sharp foreign matter that can cause injury.

The OMRR charge a qualified professional with the responsibility of evaluating sites for residual use, accurately determining application rates and minimizing the opportunity for adverse impacts on human health and the environment. The term qualified professional, means an individual who:

- is registered in British Columbia (BC) with a professional organization, is acting under that organization's code of ethics, and is subject to disciplinary action by that organization, and
- through suitable education, experience, accreditation and knowledge, may reasonably be relied on to provide advice within their area of expertise, as it relates to the regulation, duty or function.

Professionals are responsible for the preparation of a land application plan (LAP) that sets out the specific BMPs for each site, and for overseeing their implementation. A LAP addresses all aspects of residual land application, from the appropriate application rate to a post-application monitoring plan. This guideline is designed to assist professionals in sampling and analysis, in preparing LAPs and as a source of related information on the best management and use of residuals.

Notification must be given of a pending biosolids application under the requirements of the OMRR. At least 30 days before the intended land application, the generator must notify the Waste Manager in the regional Ministry of Environment (MoE) office of the proposed application and provide a subset of information required in the preparation of the LAP. Under certain land use classifications, the notification must also be provided to the Medical Health Officer having jurisdiction (MHO) over the area and to the Land Reserve Commission. Once received, the MoE and the MHO have 30 days to review the information on the proposed application and require the discharger to meet conditions, site-specific standards and management practices. The proposed application is considered acceptable if within the 30-day period:

- the MoE manager requests no further information; and,
- the MHO requests no further information and does not stop the application from proceeding. In both of the above instances neither the MoE nor the MHO approves the application; the notification is sent as a referral for review and comment as required.

United States the Environmental Protection Agency

In the United States the Environmental Protection Agency (EPA) provides leadership and has regulatory authority over the management of wastewater biosolids. In 1972, the Federal Water Pollution Control Act (Clean Water Act) required the EPA to identify biosolids disposal options (Water Environment Federation National Biosolids Partnership 2013b). As a result, in 1993, the EPA promulgated 40 CFR Part 503: Standards for the Use and Disposal of Sewage Sludge (often referred to simply as Part 503 Sludge Rule) (Water Environment Federation National Biosolids Partnership 2013b). These regulations for contaminants in biosolids were science and risk based. As a result of this approach it was determined that the limits set in the Part 503 rule should be revisited over time. The rules have been the focus of 2 National Academy of Science panels:

- The first panel completed their work in 1996 and focused on agronomic use of biosolids. This assessment found that, when the appropriate restrictions were followed, Class A and B biosolids did not pose a risk to human heath (NRC 1996). Uncertainty was expressed with respect to grazing of pastures to which Class B biosolids had been applied. It was also recommended that wastewater treatment facilities should routinely monitor and report the composition of biosolids to the public (NRC 1996).

- A 2nd panel, which reported in 2002, focused on human health. While no significant concerns were found with respect to the regulations regarding contaminants, there were recommendations that a new survey of the composition of biosolids in the US be conducted and that an updated risk assessment be performed using more modern risk assessment methods. The 2nd review panel found no documented evidence of illness related to exposure to biosolids – with the exception of a higher rate of illness in wastewater treatment plant workers (NRC 2002).

Currently, the Part 503 rule does not regulate any organic contaminants. Eighteen organic contaminants were evaluated by the EPA prior to their deletion from the final Part 503 rule based on at least one of the following justifications: use of the organic contaminants was restricted, banned, or it was no longer manufactured in the USA; the organic contaminants had a low level of detection in the National Sewage Sludge Survey (NSSS); and/or the concentration of an organic contaminants was not expected to exceed the limit based on NSSS data (NRC 1996; Smith 2009). The NRC reports question the adequacy of the NSSS and recommend that the EPA perform an updated sewage sludge survey including the measurement of a wider range of organic contaminants (NRC 1996; NRC 2002). The EPA conducted a Targeted National Sewage Sludge Survey (TNSSS) in 2006 and 2007 that was designed to: 1) obtain updated occurrence information on nine analytes of potential concern, and 2) obtain occurrence information on a number of contaminants of emerging interest identified by EPA and the National Research Council (USEPA 2009). The objective of the survey was to obtain national estimates of the concentrations of these pollutants in sewage sludge for use in assessing if exposures may be occurring and whether those levels may be of concern. A full probabilistic Risk Assessment for the Phase I TNSSS pollutants (i.e., barium, beryllium, manganese, molybdenum, silver, pyrene, 4-Chloroaniline, fluoranthene, nitrate and nitrite) was completed at the end of 2014 and circulated for review. An updated the Technical Background Document with new text that incorporates comment responses was proposed for the 3rd quarter of FY2015. In addition, screening Phase II pollutants (up to 135 pollutants with sufficient data) utilizing a newly developed screening tool and input from the 10 TNSSS pollutants risk assessment was planned for the 4th quarter of FY2015.

US EPA Reports

The US EPA Biosolids Web site (http://www.epa.gov/biosolids) contains numerous resources to support scientific, technical and regulatory aspects of biosolids management. The documents of relevance to the material covered in this report include a guides to risk assessment (USEPA 1995; USEPA 2011), emerging technologies for biosolids management (USEPA 2006), and the use of composting for biosolids management (USEPA 2002).

European Union

In the European Union the management of wastewater is guided by the Urban Waste Water Treatment Directive (91/271/EEC) and the fate of sewage sludge by the Sewage Sludge Directive (86/278/EEC). The opinion of the European Commission is that the use of sewage sludge on agricultural soils as a fertilizer is the best environmental option provided that it does not pose any threat to the environment as well as to animal and human health (Smith 2009). The Sewage Sludge Directive requires that untreated sludge be injected or incorporated into the soil. Biosolids (treated sludge) should be managed to reduce potential health risks. For example, biosolids should not be applied to fruit and vegetable crops ten months prior to harvest or pastures three weeks prior to animal grazing. The nutrient value of the biosolids should be considered when being applied to crops. The Sludge Directive is currently being considered for review.

The method of sewage sludge treatment and utilization varies considerable across the European Union (Fig. 6). The dominant method of sewage sludge treatment/utilization includes land application to agricultural lands (Spain and Portugal), composting and land application (Finland, Estonia), landfilling (Greece, Malta), and incineration (Netherlands, Switzerland) (Fig. 5; Fonts et al. 2012).

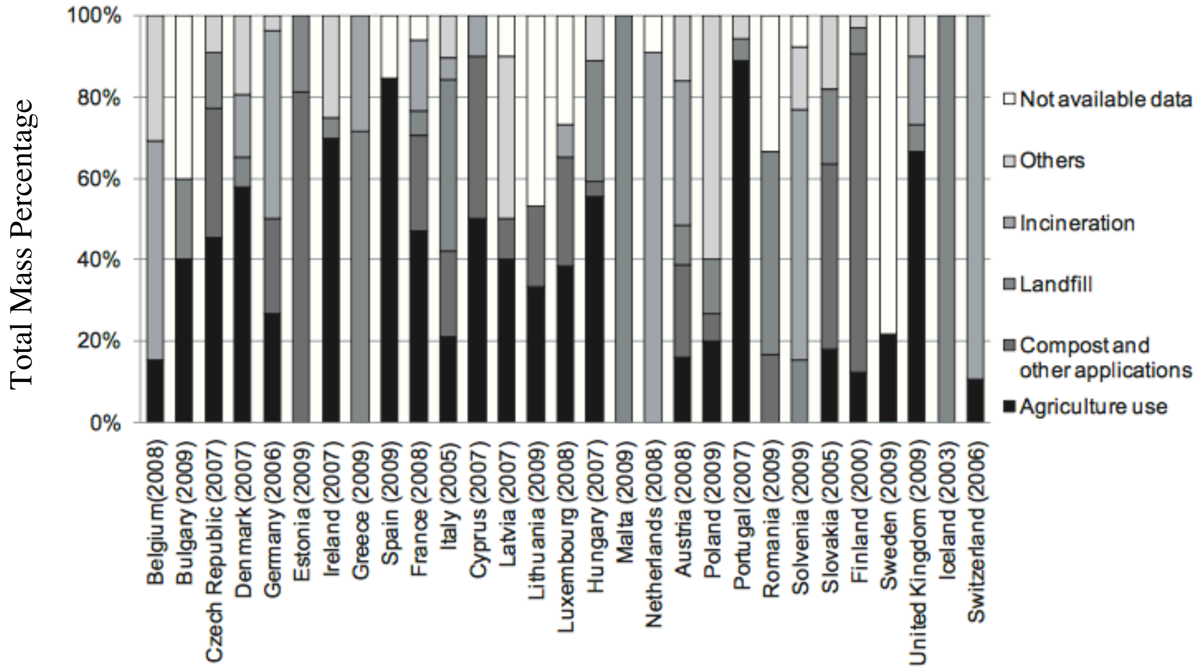

Figure 6: Sewage sludge use by type of treatment in the European Union based on latest available year of information (From Eurostat as cited by Fonts et al. 2012).

The European Commission has created an end-of-waste criterion that defines when a waste can be considered transformed into a secondary raw material and no longer a waste product. In an effort to comply with the end-of-waste criterion there is movement within the EU to increase the extent of processing of sewage using processes such as compositing and co-incineration to reduce or eliminate the potential for environmental impacts (Mininni et al. 2015). This is in part to produce a product that can be excluded from the waste directive (Directive 2008/98). The end-of-waste criterion requires that four conditions be satisfied:

a) The substance or object is commonly used for specific purposes.
b) A market or demand exists for such a substance or object.
c) The substance or object fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to product.
d) The use of the substance or object will not lead to overall adverse environmental or human health impacts.

Broadly, across the European Union, the current criteria suggest that input materials must not be "contaminated" which would exclude sewage sludge (untreated and raw sludge) (Mininni et al. 2015). Individual member states, however, may allow the use of (untreated and raw sludge) sewage sludge in this manner as a local exclusion (Mininni et al. 2015).

4. Contaminant Sources and Concentrations in Biosolids

Contaminant Sources in Biosolids

The land application of biosolids holds great potential for improvement of agricultural, rangeland, forest, or other land uses, but there is also a substantial body of evidence that shows an increasingly large number of organic contaminants are ubiquitous in municipal biosolids (Hydromantis 2009 2010; USEPA 2009). Municipal sewage treatment systems receive wastewater from industrial, commercial and residential sources, each contributing different types of contaminants to the wastewater (Hydromantis 2009 2010; Joo et al. 2015). Many organic by-products of chemical processes, solvents, and surfactants are contributed by industry (Soares et al. 2008). Households are significant contributors of pharmaceuticals and personal care products (PPCPs), cleaning products, as well as flame retardants that may be washed off fabrics (Kolpin et al. 2002). Personal care products originate from antimicrobial soaps and toothpastes, synthetic musks, sunscreen, or other products being washed off daily (La Farre et al. 2008). Active pharmaceutical ingredients are excreted either unchanged or as metabolites in human wastes, but also through dumping unused medication in the toilet (Ruhoy and Daughton 2008). Hospitals, nursing homes, dental offices, and veterinary clinics have all been identified as contributors of pharmaceuticals to municipal sewage treatment (Ruhoy and Daughton 2008).

Pharmaceutical usage in British Columbia as reported by the Canadian Institute for Health Information in 2013 is presented in Table 4, with the top 20 drug classes ranked by frequency of use. For the top drug class of extended spectrum penicillins (antibiotics) using an average daily dose of 2 g over 10 days (WHOCCDSM 2015), potentially a total of 10 Mg of this drug class is emitted to wastewater streams in BC annually. For the macrolides, another antibiotic group, potentially 1.9 Mg is added to wastewater annually in BC based on 1 g daily dose over 7 days (WHOCCDSM 2015). Antibiotics were measured at the highest concentrations (up to $\mu g\ g^{-1}$) and were among the most frequently detected compounds (3 out of 3 samples) in biosolids from Salmon Arm, BC, included in the CCME survey of emerging contaminants in biosolids (Hydromantis 2010). The presence of pharmaceuticals in biosolids is directly influenced by the use of pharmaceuticals in the population serviced by the wastewater treatment plant. Source reduction efforts may allow for reductions in the concentration of pharmaceuticals in biosolids. The British Columbia Medications Return Program, administered by the Health Products Stewardship Association, provides a means for citizens to return unused and expired medication, reducing disposal into the wastewater stream (HPSA 2014).

Recommendation 2: Source reduction initiatives should continue to be a primary focus in reducing the toxicity of materials entering the waste stream and the need for the land disposal of these compounds.

After rain events in urban centres, a large amount of contaminated water and sediment can be mobilized to enter storm water collection systems which are often connected to the municipal wastewater treatment stream (Badin et al. 2008). Polycyclic aromatic hydrocarbons (PAHs) can be present in stormwater and sediment runoff (Oleszczuk 2006), as well as pesticides, phthalates, and nonylphenols that are accumulated from water contacted surfaces (Eriksson et al. 2007). Cresols are methyl-substituted phenols that are found in wood preservatives like creosote, and have been observed in groundwater near wood treatment facilities using creosote (Rosenfeld and Plumb 1991). Cresols are also produced during wood combustion (Edye and Richards 1991) and are dispersed into the environment in smoke. Cresols may enter wastewater dissolved or sorbed to organic matter in storm runoff coming in contact with treated wood surfaces or smoke residues. Polychlorinated dibenzodioxins and furans (PCDDs and PCDFs) are produced by the incineration of chlorinated wastes, emissions from motor vehicles, and the production of other chlorinated compounds, and can enter the wastewater stream in industrial wastewater (Rogers 1996). Another potential source of contaminants to the wastewater stream is the leachate collected from modern sanitary landfills, which is sometimes treated at wastewater treatment plants. Landfill leachate can contain a wide range of organic and inorganic contaminants (Barnes et al. 2004; Choi and Lee 2006; Masoner et al. 2015).

Table 4: Top 20 Drug Classes by Rate of Use in 2013for British Columbia (CIHI 2016)

Drug Class Code	Drug Class	Rate of Use	Number of Active Beneficiaries
J01CA	Penicillins with extended spectrum	18.1%	508,959
N02AA	Natural opium alkaloids	15.9%	447,596
C10AA	HMG-CoA reductase inhibitors	15.2%	427,278
C09AA	ACE inhibitors, plain	11.6%	326,240
J01FA	Macrolides	9.8%	277,111
N06AB	Selective serotonin reuptake inhibitors	9.4%	263,353
R03AC	Selective beta-2-adrenoreceptor agonists	9.2%	259,721
H03AA	Thyroid hormones	9.0%	253,373
N05BA	Benzodiazepine derivatives	8.6%	243,142
D07AC	Corticosteroids, potent (group III)	8.5%	238,183
J01MA	Fluoroquinolones	8.0%	225,891
M01AE	Propionic acid derivatives	7.5%	210,094
J01DB	First-generation cephalosporins	7.4%	207,191
C07AB	Beta-blocking agents, selective	7.4%	207,187
H02AB	Glucocorticoids	6.9%	195,339
C03AA	Thiazides, plain	6.9%	193,339
C08CA	Dihydropyridine derivatives	6.2%	175,785
N06AX	Other antidepressants	6.2%	174,351
A10BA	Biguanides	6.1%	171,927
G03AA	Progestogens and estrogens, fixed combinations	4.9%	137,414

Concentrations of Emerging Substances of Concern (ESOCs) in Biosolids

As part of the CCME report on Emerging Substances of Concern (ESOCs) in Biosolids, a literature review was conducted to document the occurrence of a wide range of contaminants in biosolids from the year 2000 onwards, with a focus on treatment process effectiveness in lowering concentrations (Hydromantis 2009). The review considered the following classes of ESOCs: Industrial chemicals (plasticizers, pesticides, perfluorinated organic compounds, solvents, etc.); Alkylphenols and their ethoxylates; Flame retardants; Hormones, steroids and sterols; Pharmaceuticals; Personal Care Products; Certain metals (arsenic, silver selenium, mercury, etc.); Other (e.g. polyaromatic hydrocarbons, polychlorinated dioxins and furans).

A Note on Units

A common convention is to express concentrations in parts-per notation for example parts per million (ppm), parts per billion (ppb) or parts per trillion (ppt). These are often used as they are more easily understood by the general public. For example:

- ug g^{-1} is one part per million is like 1 metre in 1,000 kms (distance from Vancouver to Saskatoon)
- ng g^{-1} is one part per billion is like 1 metre in 1,000,000 kms (distance from Vancouver to the moon three times)
- pg g^{-1} is one part per trillion is like 1 metre in 1,000,000,000 kms (distance from Vancouver to almost Saturn)

There is an interesting and informative YouTube presentation on the ***Powers of Ten*** that also provides an effective illustration to the magnitude of these units.

These are not, however, strictly speaking SI units. They are also somewhat vague in that it is not clear whether they refer to mass fraction (e.g., ug g^{-1}), mole fraction (e.g., umole $mole^{-1}$), volume fraction (e.g., uL L^{-1}) or a mixture of these units (e.g., mg L^{-1}). For clarity in this report we have chosen to represent the actual mass fraction or volume fraction units rather than parts-per notation to avoid confusion.

While fate during sewage treatment and occurrence in biosolids has been extensively characterized for some compounds (e.g. bis(2-ethylhexyl)phthalate, nonylphenol and some ethoxylates, galaxolide, tonalide, polybrominated diphenyl ethers, and triclosan), there is a lack of information available for many antibiotics and other pharmaceuticals, parabens (antimicrobial preservatives), sunscreen agents, and insect repellents (Hydromantis 2009). Table 5 lists the approximate range of concentrations for different contaminants or classes reported in biosolids based on the literature review (Hydromantis 2009). Most contaminants were in the ng g^{-1} to μg g^{-1} range, with exceptions being linear alkylbenzene sulfonate (LAS) and the plant and animal sterols in the mg g^{-1} range. Plant and animal sterols are naturally occurring and may serve as anthropogenic indicators, while data on LAS was dated (1990s) and concentrations may now be lower, although there is a lack of recent occurrence data (Hydromantis 2009).

Table 5: Range of typical concentrations of contaminant classes or individual contaminants in biosolids reported in the literature, taken from the CCME Final Report – Literature Review on ESOCs in biosolids (Hydromantis 2009).

Contaminant or Class of Contaminant	Concentration Range in Biosolids		
	ng g^{-1}	$\mu g\ g^{-1}$	mg g^{-1}
Bisphenol A (in plastic, epoxy, thermal receipt paper)	■	■	
Phthalates (in plastics)	■	■	
PFOCs (non-stick coatings)	■		
LAS (laundry detergents)		■	■
Chlorophenolics (pesticides, disinfectants)	■	■	
Pesticides (lawn and garden, agriculture)	■	■	
Solvents (industrial cleaners, degreasers)	■		
Alkylphenols (cleaning agents)		■	
PBDEs (flame retardants)	■		
Pharmaceuticals (prescription, over-the-counter)	■	■	
Estrogens and androgens (hormonal medications)	■		
Plant Sterols (feces)		■	■
Animal Sterols (feces)		■	■
Triclosan (toothpaste, antimicrobial soap)		■	
Musk Fragrances (perfumes, colognes, body wash)	■	■	
Quaternary Ammonium Compounds (industrial sanitizers)		■	
Siloxanes (silicone)		■	
Polycyclic Aromatic Hydrocarbons (combustion by-products)	■	■	
Polychlorinated polyaromatics (combustion by-products)	■		

The CCME report on ESOCs in biosolids also involved a nation-wide field sampling program from eleven WWTPs across Canada, to quantify the magnitude and frequency of occurrence of 57 pharmaceutical compounds, 3 alkylphenolic compounds (including Bisphenol A), 11 synthetic musk fragrances, and 11 metals (discussed in Metals section). A summary of compounds detected, the frequency of detection, and median detected concentrations measured are presented in Table 6. The number of pharmaceuticals detected in 75% of samples decreased from 20 to 10 between the feed sludge and finished product samples, suggesting treatment can reduce the number of contaminants detected in biosolids, although any reductions are dependent on the treatment process and specific contaminants considered (Hydromantis 2010). A small proportion (12 out of 57) of pharmaceuticals were measured at > 1 $\mu g\ g^{-1}$ in finished biosolids, with triclosan, triclocarban, and ciprofloxacin exceeding 1 $\mu g\ g^{-1}$ at 9 out of 11 sites (Hydromantis 2010). HHCB, AHTN (fragrances), and Bisphenol A (plasticizer) exceeded 1 $\mu g\ g^{-1}$ at 10, 6, and 3 out of 11 sites respectively (Hydromantis 2010). While some compounds are only detected sporadically, others are essentially ubiquitous in biosolids from Canadian WWTPs, including HHCB, triclocarban, AHTN, miconazole, diphenhydramine, carbamazepine, triclosan, ATII, and ciprofloxacin, which were detected in >90% of samples analyzed and with median concentrations in the high ng g^{-1} to low $\mu g\ g^{-1}$ range (Table 4). For context, many of the compounds occur at much higher concentrations in regulated common use. For example, triclosan is present at 3,000 $\mu g\ g^{-1}$ in certain toothpaste products, although certain aroma compounds like sotolon, the main component of fenugreek seed used in curry powders and one of the components of artificial maple syrup, has an odour detection threshold in water of about 300 ng g^{-1}, or parts per billion, in water (Marsili 2011).

Table 6: Occurrence of ESOCs in samples of biosolids produced from eleven WWTPs across Canada, as reported in the CCME Final Report – Field Sampling Program on ESOCs in biosolids (Hydromantis 2010).

Compound	% occurrence	Median of detected conc. (ng/g TS dw)	Compound	% occurrence	Median of detected conc. (ng/g dw)
HHCB	100%	3470	Gemfibrozil	52%	56
Triclocarban	100%	1930	Trimethoprim	42%	31.2
AHTN	100%	1340	Dehydronifedipine	42%	7
Miconazole	100%	441	Sulfamethoxazole	39%	5.2
Diphenhydramine	100%	420	Furosemide	32%	543
Carbamazepine	100%	66.6	2-Hydroxy-ibuprofen	26%	497
Triclosan	97%	6085	Enrofloxacin	23%	22.2
ATII	96%	255	Octylphenol	18%	50
Ciprofloxacin	94%	3610	1,7-Dimethylxanthine	13%	378
Ofloxacin	87%	276	Sulfanilamide	13%	63.1
Bisphenol A	86%	325	Glyburide	13%	11.5
Azithromycin	84%	205	Hydrochlorothiazide	10%	143
Fluoxetine	84%	53.9	Sulfamerazine	10%	17.9
Naproxen	81%	98.1	Virginiamycin	6%	197
Clarithromycin	74%	41.8	Digoxin	6%	192
Thiabendazole	74%	17.9	Digoxigenin	6%	128
Erythromycin-H_2O	74%	12.5	Musk Xylene	5%	530
DPMI	73%	82.5	ADBI	5%	60
Ibuprofen	68%	522	Lincomycin	3%	71.1
Diltiazem	68%	29.8	Penicillin V	3%	59.3
AHDI	64%	158	Glipizide	3%	11.4
Caffeine	61%	266	Oxolinic Acid	3%	1.9
Norfloxacin	58%	558	Roxithromycin	3%	0.8

Concentrations of Metals in Biosolids

The CCME report field sampling program on ESOCs in biosolids included 11 metals analyzed on a single sample of biosolids from each site. Concentrations detected across the eleven different biosolids are summarized in Table 5, compared to limits for Biosolids Growing Medium or Class B Biosolids from the BC Organic Matter Recycling Regulations, and the percent reduction in metals concentration compared to historical biosolids measurements (Hydromantis 2010). With the exception of arsenic, there was a reduction in metals concentration since either 1981 or 1995 (depending on the availability of data). Reductions ranged from 35% for selenium to 98% for chromium. Metal concentrations for biosolids, compost and soil are regulated, and metal quantities in biosolids have been effectively controlled through source control, pre-treatment, and sewer use limits (Hydromantis 2010). The median concentration in detected samples for copper, selenium, and zinc exceeded established limits for compost, while chromium, mercury, and molybdenum limits were exceeded by the maximum detected concentration in the samples tested (Table 7). Biosolids treatment processes are unable to remove metals, so reductions in biosolids concentration must come through source reduction (Hydromantis 2010).

Table 7: Metals concentration in Canadian biosolids as measured by (Hydromantis 2010) in comparison to the limits for Biosolids Growing Medium or Class B Biosolids from the BC Organic Matter Recycling Regulations, and the percent reduction in metals compared to 1981 or 1995 levels. Bold concentrations exceed the OMRR for a biosolids growing medium.

Metal	No. of Detected Conc. (out of 11)	Concentration (mg/kg TS dw)			OMRR Limit for Biosolids Growing Medium	OMRR Limit for Class B Biosolids	% Reduction (Current compared to 1981 or 1995)
		Median of All Conc.	Median of Detected Conc.	Maximum Detected Conc.			
Arsenic	7	1.4	2.6	6.7	13	75	-13%
Cadmium	2	<1.0	1.1	1.2	1.5	20	97%
Chromium	10	18.1	20.3	**120**	100	1060	98%
Cobalt	7	2.6	2.9	4.2	34	150	NA
Copper	11	**271**	**271**	**890**	150	2200	69%
Lead	9	22.5	24.7	55.5	150	500	96%
Mercury	11	0.68	0.68	**3.2**	0.8	15	81%
Molybdenum	8	1.8	3.5	**8.6**	5	20	84%
Nickel	9	9.9	10.5	21.1	62	180	93%
Selenium	6	1.3	**2.2**	**3.2**	2	14	35%
Zinc	11	**331**	**331**	**647**	150	1850	76%

For context, animal manures measured in England and Wales had metal contents (in mg kg^{-1}) in the following ranges: As 0.46 to 9.01; Cd 0.13 to 1.06; Cr 1.41 to 17.17; Cu 16.4 to 374; Pb 1.95 to 8.37; Ni 2 to 10.4; Zn 81 to 575 (Nicholson et al 1999). Values were not reported for cobalt, mercury, molybdenum, or selenium. With the exception of copper and zinc, the maximum measured concentrations were all lower than the OMMR limit for biosolids growing medium.

Contaminant Fate in WWTPs and the Effect of Biosolids Treatment

The fate of individual compounds in sewage treatment depends on the physiochemical properties of the compound, such as lipophilicity, polarity, water solubility, and vapour pressure (La Farre et al. 2008). Organic contaminants tend to accumulate in sludge owing to their lipophilic and hydrophobic properties and consequently they sorb onto the sludge organic matter (Smith 2009a). For example, a large proportion of *p*-cresol is expected to be sorbed to organic matter during sewage treatment, since cresols are slightly soluble in water (Lide 2005) and have an octanol-water partition coefficient (log k_{ow}) of about 1.9 (Xie et al. 2008). Cresols have been found in biosolids destined for land application (Kinney et al. 2006) and in soils receiving biosolid amendments (Burkhardt et al. 2005; Kinney et al. 2006). Oleszek-Kudlak et al. (2005) studied the fate of PCDDs and PCDFs in a WWTP, and found some compounds increased while others decreased in concentration between the incoming and outgoing water. Increases were attributed to transformations such as dechlorination by microbes to yield lower chlorinated compounds, while decreases were expected due to sorption or degradation (Oleszek-Kudlak et al. 2005). Mass balances on PBDEs in a number of WWTPs suggest that over

95% ends up in the sludge (Peng et al. 2009; Ricklund et al. 2009). The contribution of BDE-209 to the congener profile decreased from influent to effluent water, while the contribution of less substituted congeners increased. The antimicrobial compound triclosan has generally high removal rates during wastewater treatment (>90%), although it is not known with a high degree of certainty if sorption or degradation is responsible for the removal (Heidler and Halden 2007; Lozano et al. 2010). Triclosan is only slightly soluble in water (12 mg L^{-1}) and hydrophobic (log k_{ow}=4.8, log k_{oc}=3.8 to 4.0), which suggests that sorption to organic matter in wastewater treatment strongly affects removal efficiency (Lozano et al. 2010).

The CCME Field Sampling Program (Hydromantis 2010) compared ESOC removal efficiency for the different biosolids treatment processes included in the study. To achieve a broad comparison, a score was calculated for each contaminant detected, ranging from 1 if 100% of the compound is removed, to 5 if there is a greater than -50% removal rate (50% increase) between raw and finished product (Hydromantis 2010). The total score and average score per compound detected are presented in Table 8. At a broad level, composting appears to provide the best reduction in contaminant concentrations as a treatment option for biosolids, since the top three performing locations all used composing. However, there are substances (e.g. naproxen) that persist and even increase in concentration due to composting (Hydromantis 2010). With the exception of the

Table 8: Effect of biosolids treatment process based on the score calculated by (Hydromantis 2010) considering removal efficiency during treatment for the number of compounds detected.

Location	Treatment Process Assessed	Score total	Number of compound (counts)	Processing /Reduction efficiency (average score)
Gatineau Valley	Biological – aerobic (Compost)	49	27	1.81
Moncton	Biological – aerobic (Compost)	57	31	1.84
Prince Albert	Biological – aerobic (Compost)	72	29	2.48
Eganville (Septage)	Physical – geotextile bag dewatering	85	28	3.04
Halifax N-Viro	Physical-chemical (alkaline stabilisation)	115	35	3.29
Red Deer	Biological – mesophilic anaerobic digestion	115	34	3.38
Salmon Arm	Biological – autothermal aerobic digestion	111	32	3.47
Saskatoon	Biological – mesophilic anaerobic digestion	118	34	3.47
Smiths Falls	Physical – thermal drying	101	27	3.74
Gander	Physical – filter press dewatering	102	27	3.78
Saguenay	Physical – filter press dewatering	108	27	4.00

septage sample (Eganville), physical treatment processes (drying, dewatering) were least effective in removing contaminants, although alkaline-stabilization (a physical-chemical process) was among the more effective treatments. Anaerobic or aerobic digestion were not as effective as composting or alkaline stabilization for decreasing contaminant concentrations, which may be dependent on the retention time during processing but the reason is currently unknown (Hydromantis 2010). As shown in Table 9, reproduced from Hydromantis (2010), the removal efficiency varies depending on both the substance and on the treatment process. In the case of triclosan (excluding the septage sample), removal efficiencies range from 99% in composting to -91% in autothermal aerobic digestion (Table 7).

Knowledge Gaps and Opportunities for Improvement

Metal concentrations in biosolids have generally declined in the last two to three decades, largely as a result of increased regulation of concentrations and management practices. In biosolids measured across Canada, the median metal concentrations were lower than the OMRR limits for plant growing medium, with the exception of copper, selenium, and zinc. Reductions in these metals may come through better identification of sources to reduce inputs to the wastewater stream.

There is conclusive evidence that a wide range of organic contaminants are present in biosolids. A number of contaminants, including HHCB, triclocarban, AHTN, miconazole, diphenhydramine, carbamazepine, triclosan, ATII, and ciprofloxacin, have a high frequency of detection (>90%) in biosolids across Canada. Biosolids treatment processes can decrease concentrations of certain contaminants, although some contaminants persist or increase in concentration. There are gaps in the understanding of removal mechanisms for the different treatment processes, and in the mechanisms leading to increased concentrations of specific contaminants after treatment. There is a need to further study the highest performing treatment processes (composting and alkaline stabilization) to optimize the removal of organic contaminants in biosolids destined for land application. There is also room for strengthened regulations with respect to monitoring organic contaminants in biosolids across Canada, to increase understanding of the scope of the issue and to inform strategies for targeted source reductions in substances that are identified as posing a risk to human health or the environment.

Recommendation 3: An organic contaminant monitoring program should be established for biosolids being applied in the Nicola Valley. This should include monitoring of concentrations in source materials and treated products.

Table 9. Removal efficiencies for detected pharmaceuticals by biosolids treatment process as reported in (Hydromantis 2010). Negative values denote an increase in concentration between the raw and finished biosolid.

Pharmaceutical	% Removal										
	Biological-autothermal aerobic digestion	Biological-aerobic (Compost)	Biological-aerobic (Compost)	Biological-aerobic (Compost)	Biological-mesophilic anaerobic digestion	Biological-mesophilic anaerobic digestion	Physical-filter press dewatering	Physical-filter press dewatering	Physical-geotextile bag filter dewatering	Phys-chem (alkaline stabilis'n)	Thermal Drying
	Salmon Arm	Moncton	Gatineau Valley	Prince Albert	Saskatoon	Red Deer	Gander	Saguenay	Eganville (Septage)	Halifax N-Viro	Smiths Falls
Furosemide	-73%			-187%				-41%	-123%	-110%	20%
Gemfibrozil	-245%	>80%	>94.2%-98.8%	55%	-56%	-168%				-25%	-17%
Glyburide			>94.9%-99.5%							>43%	
Hydrochlorothiazide		>55%			36%	-79%			>75%- 80%	39%	
2-Hydroxy-ibuprofen	-73%		>46.7%-97.7%		3%	61%			>35%-37%	-5%	
Ibuprofen	-306%	>77%	95%	16%	-82%	-15%	-30%		13%	-82%	-8%
Naproxen	-141%	-2837%	-4270%	-1424%	62%	60%	-6%	-429%	23%	-17%	-11%
Triclocarban	-11%	95%	95%	49%	-20%	-39%	6%	2%	64%	53%	9%
Triclosan	-91%	95%	99%	68%	-55%	-17%	17%	21%	-123%	12%	1%
Acetaminophen		>38%			>55.2%		>-85.1%		78%		
Azithromycin	-6%	>98%	98%	93%	-12%	7%	4%	11%	14%	88%	-34%
Caffeine	-141%	>97%	>98.0%-99.6%	6%	92%	94%	4%	-290%	-26%	25%	-66%
Carbamazepine	-150%	89%	45%	50%	-112%	-219%	-40%	-35%	-183%	36%	-108%
Ciprofloxacin	61%	94%	96%	82%	-6%	-25%	1%	9%	-228%	44%	45%
Clarithromycin	-32%	>94%	>41.2%-99.9%	81%	43%	54%	-27%	-55%	-10%	59%	19%
Clinafloxacin										>-50%	
Dehydronifedipine	51%	>54%	67%	>79%	>13.3%	>19%	-149%			-9%	-121%
Diphenhydramine	-1%	99%	96%	74%	12%	-18%	-2%	-1%	36%	76%	-45%
Diltiazem	92%	>99%	>99.1-99.8%	93%	89%	86%	14%	-90%	61%	>76%	-94%
Digoxin						>64%					
Digoxigenin						37%					
Enrofloxacin		>-13%			-57%	49%		-36%		>-1%	22%
Erythromycin-H_2O	14%	>93%	>31.2%-98.8%	86%	41%	54%	-40%	-68%	-239%	55%	-1%
Fluoxetine	43%	>82%	91%	60%	40%	-39%	-49%	-64%		56%	-10%
Lincomycin						>13%					
Miconazole	-47%	95%	95%	64%	-97%	-113%	>3.6%	-16%	<-65%	32%	-45%
Norfloxacin	72%	>88%			71%	-31%	-23%	-7%	<-762%	-5%	46%
Ofloxacin	44%	56%	43%	73%	-6%	-44%	>-4.5%	-10%		-49%	39%
Oxolinic Acid				>-20%							
Sulfamerazine					-17%						
Sulfamethazine				>75%		>19%					
Sulfamethoxazole	90%	>65%		>98%	>90.4%	>77%	12%	-26%	77%	-19%	-282%
Sulfathiazole				>48%							
Thiabendazole	-1%	>72%	>97.2%-99.9%	70%	-40%	-69%	-22%	-21%		-28%	4%
Trimethoprim	55%	>90%		81%	91%	>79%	-4%	10%	-9%	7%	-3%
Virginiamycin										-47%	
1,7-Dimethylxanthine	-34%		>96.6%-99.1%						>63%-65%	19%	

5. *Risk Factors for Organic Contaminants in Land-Applied Biosolids*

The World Health Organization (WHO) defines a risk factor as "any attribute, characteristic or exposure of an individual that increases the likelihood of developing a disease or injury" (WHO 2016). The risk of human or environmental harm from a contaminant in biosolids is dependent on a number of factors that influence the exposure pathways leading to a detrimental effect on an organism. There are three main risk factors that determine the risk from organic contaminants in the environment: Persistence, Bioaccumulation, and Toxicity (PBT) (Arnot and Gobas 2006; Clarke and Cummins 2015; Government of Canada 1999 2000; USEPA 1999). Biosolid contaminants that are toxic pose a greater hazard when they persist for longer periods in the environment, and when they bioaccumulate in exposed organisms (Clarke and Cummins 2015). Persistence and bioaccumulation depend on the chemical structure and properties (e.g. sorption coefficient K_d, octanol-water partition coefficient log k_{ow}, acid dissociation coefficient pKa) of the substance, as well as the properties of the environmental matrix (e.g. soil texture, pH, organic matter content, cation exchange capacity), with climatic variables of temperature and rainfall exerting influence at larger spatial scales (Verlicchi and Zambello 2015b). The PBT risk factors are defined and discussed in more detail in the next three sections.

Toxicity

The Canadian Environmental Protection Act (Government of Canada 1999) states that a substance is toxic if it enters the environment in a concentration or quantity that will "(a) have or may have an immediate or long term harmful effect on the environment or its biological diversity; (b) constitute or may constitute a danger to the environment on which life depends; or (c) constitute or may constitute a danger in Canada to human life or health". The USEPA (1999) definition of toxicity is similar in effect, although it lists specific human health impacts, including cancer, teratogenic or reproductive effects, genetic mutations, neurological disorders, or other chronic health problems. The EU definition of toxicity requires a substance to meet any of the following situations: (a) the substance has a long-term EC_{10} or NOEC < 0.01 mg L^{-1} for freshwater or marine organisms, (b) the substance meets the EU criteria for carcinogenic, germ cell mutagenic, or toxic for reproduction, (c) there is evidence of chronic toxicity for the substance based on criteria for repeated exposure to target organs (European Commission 2011).

Assessment of toxicity to an organism involves determining the effective dose or concentration at which specific impacts occur (Calow and Forbes 2003). Human health risk assessments consider cancer and non-cancer based endpoints, as well as loss of life years due to illness (NHRMC 2008; Schriks et al. 2010). Ecotoxicity end-points range from the extreme of mortality to subtler impacts at sub-lethal concentrations, including decreased growth or reproduction, genetic damage, or expression of biochemical markers of stress (Bundy et al. 2009; Lin et al. 2010; Spurgeon et al. 2003). Determining the environmental concentration or dose at which a substance is toxic and causes harm to an organism is critical in assessing risk, as it forms the threshold to which measured or predicted exposure concentrations are compared (European Commission 2003).

Persistence

The Persistence and Bioaccumulation Regulations (Government of Canada 2000) issued under the Canadian Environmental Protection Act (Government of Canada 1999) state that a substance is persistent if it (a) has a half-life in air ≥ 2 days or can be transported atmospherically to a remote location, or (b) has a half-life ≥ 182 days in water or soil. The half-life is defined as "the period it takes the concentration of a substance to be reduced by half, by transformation, in a medium" (Government of Canada 2000). The USEPA (1999) criteria for persistence are lower for soil and water with half-lives of > 2 months, and the same in air (> 2 days). The rationale for their decision is that persistence is a continuum, and that a substance with a half-life of > 6 months is highly persistent, while a half-life between 2 to 6 months is still persistent to some degree (USEPA 1999). The European Commission definition of persistence classifies substances as either persistent or very persistent (European Commission 2011). Persistent substances are defined as having a half-life greater than the following values in the matrices listed: (a) marine water – 60 days; (b) estuarine water – 40 days; (c) marine sediment – 180; (d) fresh or estuarine water sediment – 120 days; (e) soil – 120 days. Very persistent substances have a half-life that exceeds any of the following half-life criteria: (a) marine, fresh, or estuarine water – 60 days; (b) marine, fresh or estuarine water sediment – 180 days; (c) soil – 180 days.

Persistence is inversely related to the ability of a substance to be degraded by both abiotic and biotic processes. Phototransformation of certain photoreactive substances like triclosan is an important process under aqueous conditions, reducing the persistence of these compounds in the photic zone of water bodies (Latch et al. 2005). However, in the case of triclosan, the transformation products can include chlorinated phenols or dioxins that have well established toxic effects (Latch et al. 2005). Geochemical transformation processes like oxidation-reduction (in the presence of oxygen, free electrons, or H^+ ions), hydrolysis, or condensation reactions with organic matter in soil can also reduce the persistence of certain contaminants in environmental matrices (Berkowitz et al. 2014).

Microorganisms can easily degrade substrates that contain functional groups familiar to all life forms, including carboxylic acids, alcohols, amines, amides, alkanes, cycloalkanes, and phenolic rings. However, many emerging substances of concern (especially pharmaceuticals) contain functional groups or chemical structures that are foreign to most or all life forms and are termed xenophores (Hickey 2005). These groups are toxic to many organisms (e.g. commercial biocides) or have unique medical applications (e.g. pharmaceutical drugs), but they impede the microbial biodegradation of chemicals in soil or water and increase persistence. They include cyano, nitro, halogen, and sulfonic acid functional groups, as well as branched alkanes and ring structures containing O, N, or S (Hickey 2005). Biodegradation first requires the removal or modification of these groups before the compound can be further transformed into a substrate that can be utilized, requiring enzymes capable of detoxifying these xenophores (Hickey 2005).

Bioaccumulation

The Persistence and Bioaccumulation Regulations (Government of Canada 2000) state that a substance is bioaccumulative if (a) it has a bioaccumulation factor (BAF) > 5000, or (b) a bioconcentration factor (BCF) > 5000 if BAF is not available, or (c) log k_{ow} > 5, if BAF or BCF are not available. The BAF is defined as "the ratio of the concentration of a substance in an organism to the concentration in water, based on uptake from the surrounding medium and food" (Government of Canada 2000). The BCF is defined similarly, but is "based only on uptake from the surrounding medium" (Government of Canada 2000). The criteria for bioaccumulation used by the (USEPA 1999) is a BCF or BAF > 1000, with the rationale for a smaller value being that bioconcentration is a continuum and substances with a BCF < 5000 are still bioaccumulative. The European Commission definition of bioaccumulative classifies substances as either bioaccumulative or very bioaccumulative (European Commission 2011). Bioaccumulative substances are defined as having a BCF > 2000 for aquatic species, while very bioaccumulative substances have a BCF > 5000.

Bioaccumulation in Fish

Bioaccumulation of a chemical in an organism is related to the rate of uptake and the rates of elimination and dilution of a chemical by various routes that are specific to the exposed organism (Arnot and Gobas 2006). In fish, the major routes of uptake are through dietary consumption and through the gills, while major elimination routes are through the gills, fecal elimination, and metabolic transformations, with a dilution of whole-organism concentration occurring due to growth (Fig. 7). Dilution due to growth is not considered a true elimination process since the total amount of a substance remains the same, but an increase in mass will lead to a lower concentration of that substance in an organism. Bioconcentration pathways are similar, with the exception that dietary uptake is not included (Arnot and Gobas 2006). Bioaccumulation or bioconcentration can be calculated on a wet or dry basis, or can also be expressed based on the lipid content of an organism which can give a better representation of lipophilic chemicals, as these chemicals tend to be concentrated in fatty tissues. For chemicals that are more strongly associated with the protein fraction, it can be more useful to express concentrations on a protein weight basis (Arnot and Gobas 2006). When expressed on a wet weight basis, the units for BCF or BAF are L kg^{-1} (Arnot and Gobas 2006).

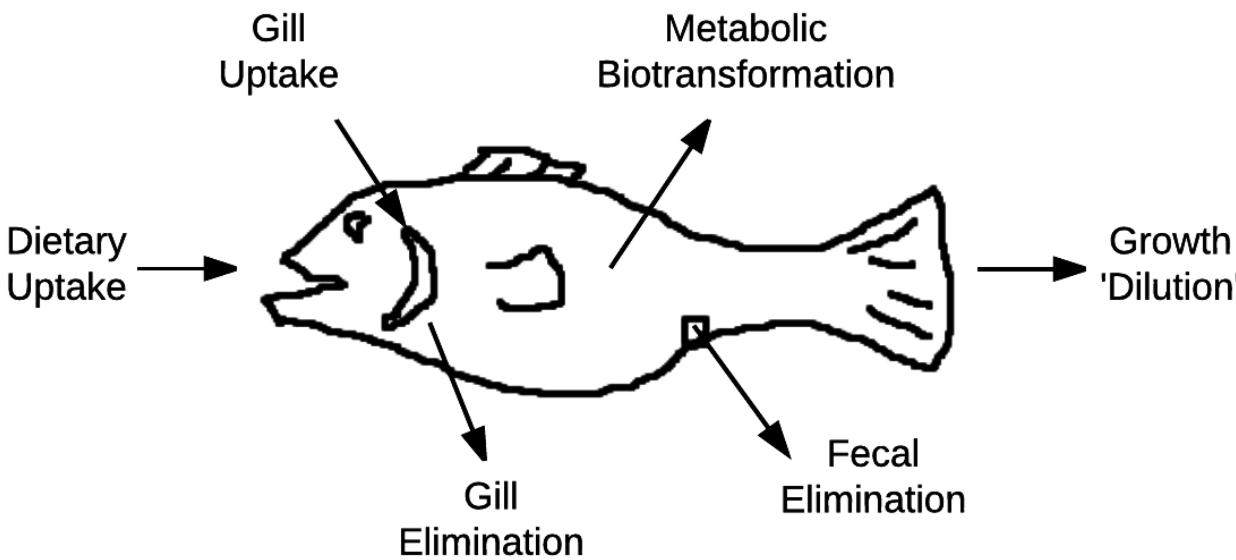

Figure 7: Major pathways of uptake and elimination that determine bioaccumulation of a chemical in fish, adapted from Arnot and Gobas (2006).

Bioaccumulation in Earthworms

Earthworms have been used in bioassays of contaminant bioaccumulation in soil, due to their physical contact with and ingestion of large amounts of soil during their lifetime (Lanno et al. 2004). However, the assessment is complicated by behavioural differences between species, and by artifacts of laboratory assays using homogenized and sieved soil that do not reflect field conditions (Jager et al. 2005). Bioaccumulation in earthworms is dependent on the desorption of contaminants from the soil matrix into pore water, which is influenced by sorption and degradation processes (Carter et al. 2014, Lanno et al. 2004) and can lead to different patterns of bioaccumulation resulting from low or high desorption rates (Jager et al. 2005). Elimination of a compound occurs if the earthworms are moved to fresh material, although tissue concentrations do not necessarily become zero since the substance leaving contaminates the soil (Jager et al. 2005). The bioaccumulation and elimination processes have been described by first order kinetics, with the rate of change in organism concentration equal to the rate of intake from pore water minus the rate of excretion of accumulated substance (Carter et al. 2014). Other approaches have used modifications to account for additional processes affecting organism concentration, such as distinguishing between passive elimination and biotransformation (Ma et al. 1998). Some authors have calculated BAFs by measuring contaminant concentrations in soil and in earthworm tissues from agricultural sites amended with both biosolids and animal manure (Kinney et al. 2008).

Bioaccumulation in Plants

The uptake of organic contaminants into agriculturally relevant plant tissues is an emerging issue receiving increased attention, with at least four review papers published on the subject in the last five years (Carvalho et al. 2014; Miller et al. 2015; Wu et al. 2011; Wu et al. 2015). There is no consensus in the scientific literature as to the best or most appropriate model describing the uptake of organic contaminants into plants, although several models can provide adequate descriptions of the process for some substances (Polesel et al. 2015; Prosser et al. 2014). Uptake of organic contaminants into plants is described by mass flow (advection) of contaminated water from soil pores into roots which is driven by transpiration, and the physicochemical properties of the compound affect the mobility of a compound within the plant (Prosser et al. 2014; Wu et al. 2015). Transport into the plant and further movement into edible tissues can be described by partition coefficients between different compartments (Prosser et al. 2014). Soil pore water concentrations are dependent on the soil concentration, but are influenced by sorption to soil organic matter and by abiotic or biotic degradation (Wu et al. 2015). Addition of biosolids to soil increases soil organic matter and consequently reduces the mobility of some pharmaceuticals in soil; however, it can also retard pharmaceutical degradation (Wu et al. 2015).

Most studies to estimate plant bioconcentration factors have been completed under laboratory or greenhouse conditions (Wu et al. 2015). Bioconcentration factors measured for plant roots and leaf/stem tissues in potting mix vary widely in magnitude based on the compound, and suggests that some compounds concentrate in roots (e.g. triclocarban, triclosan) while others (e.g. carbamazepine, fluoxetine, diazepam) have higher potential to be transported within a plant (Wu et al. 2015). For plants grown in soil, the BCFs were generally lower than those measured for potting mix. Estimates for human exposure to PPCPs from consumption of contaminated plant tissues are generally <1% of a medical dose or acceptable daily intake (ADI), although there is limited evidence that triclosan, lamotrigine, and 10-11-epoxycarbamazepine can approach or exceed acceptable limits (Wu et al. 2015). The exposure of humans to PPCPs through the diet is estimated to be low, although there are specific contaminants of concern and a comprehensive human health risk assessment has not been completed, particularly concerning exposure to multiple PPCPs (Wu et al. 2015).

Bioaccumulation in Meat, Milk, and Higher Organisms

Dietary consumption of contaminated food is not just a concern for the directly-exposed organisms. In a food chain, bioaccumulation in lower trophic level organisms can lead to increasing chemical concentrations at higher trophic levels through a process called biomagnification, with the biomagnification factor (BMF) defined as the ratio of concentration of a substance in an organism to the concentration in its diet (Arnot and Gobas 2006). The Biotransfer Factor (BTF) is a similar term used to describe the transfer of a chemical into livestock or other animals from the consumption of contaminated food, and is defined as the ratio of a chemical concentration in meat or milk (mg kg^{-1}) to the chemical intake rate (mg day^{-1}), with the units of day kg^{-1} (USEPA 2005). Since lipophilic compounds accumulate in fat, the fat-adjusted BTF can be estimated from the log K_{ow} of the substance if empirical data is unavailable (USEPA 2005).

Knowledge Gaps and Opportunities for Improvement

Environmental fate is frequently predicted based on physico-chemical properties of the substance, and empirical validation is required to verify the accuracy of mass balance predictions. Knowledge gaps have been identified surrounding the plant uptake of organic contaminants, including relationships between the soil fate of PPCPs and the chemical and soil properties which are incomplete, poor understanding of the mechanisms and pathways of PPCP degradation in soil, limited plant uptake data availability under field conditions, poor understanding of ionisable substances, low availability of isotopically labelled standards for PPCPs and their metabolites, and a lack of understanding of the potential effects of exposure to low doses of multiple PPCPs and their transformation products in humans (Wu et al. 2015).

6. *Review of Risk Assessments on Organic Contaminants in Biosolids*

Overview of Risk Assessment

Risk assessments vary in complexity, but generally consider a number of exposure pathways (e.g., Biosolids –> Soil –> Plant –> Human, Figure 7) whereby an estimate of exposure dose by a pathway is calculated or measured, and compared to a reference dose for which a toxic effect has been determined. The resulting ratio between the exposure and effects concentration (termed the Hazard Quotient, Hazard Index, Risk Quotient, or other similar names) is an index of risk. A value of ≥1 means that the exposure concentration meets or exceeds the concentration where a known toxic effect is observed, presenting a high risk for that exposure pathway, while values between 0.1 and 1 indicate moderate risk and between 0.01 to 0.1 a low risk (Hernando et al. 2006).

In ecotoxicology, environmental concentrations can be considered analogous to the exposure dose for estimating the dose-response relationship for toxic effects and calculating the predicted-no-effect concentration (PNEC) or 50% effective concentration (EC_{50}) values (Calow and Forbes 2003). The PNEC or EC_{50} is compared to measured or predicted environmental concentrations (MEC or PEC) to determine risk. If no soil-based toxicity values are available for a compound, the PNEC or EC_{50} can be estimated from aquatic toxicity values based on the solid-water distribution coefficient (K_d) using the equilibrium partitioning method (European Commission 2003; Verlicchi and Zambello 2015). There can be considerable (order of magnitude) variation in calculated values depending on the source of the K_d values and the extent to which they represent the matrix being studied (McCarthy et al. 2015; Verlicchi and Zambello 2015).

For screening purposes, factors of 10-1000 can be applied to account for uncertainty in species sensitivity when extrapolating toxicity indicators (such as PNEC or EC_{50}) across different species when data availability is low (Calow and Forbes 2003; European Commission 2003). In conducting a risk assessment, it is assumed that the sensitivity of an ecosystem to a toxic compound is dependent on the most sensitive species, and that protecting the most sensitive species protects all less sensitive species and thereby protects the structure and function of an ecosystem (European Commission 2003). As the number of estimates for the toxicity threshold of a substance increases, a smaller uncertainty factor is used to account for greater certainty in the range of threshold values, and the toxicity threshold of the most sensitive species is used for risk assessment (European Commission 2003). If empirical exposure data is unavailable but persistence and bioaccumulation data are known, risk assessments can consider the predicted dose of a compound delivered to an exposed population based on mass balance models for transfers within the exposure pathways depicted in Figures 7 and 8 (European Commission 2003; Verlicchi and Zambello 2015b).

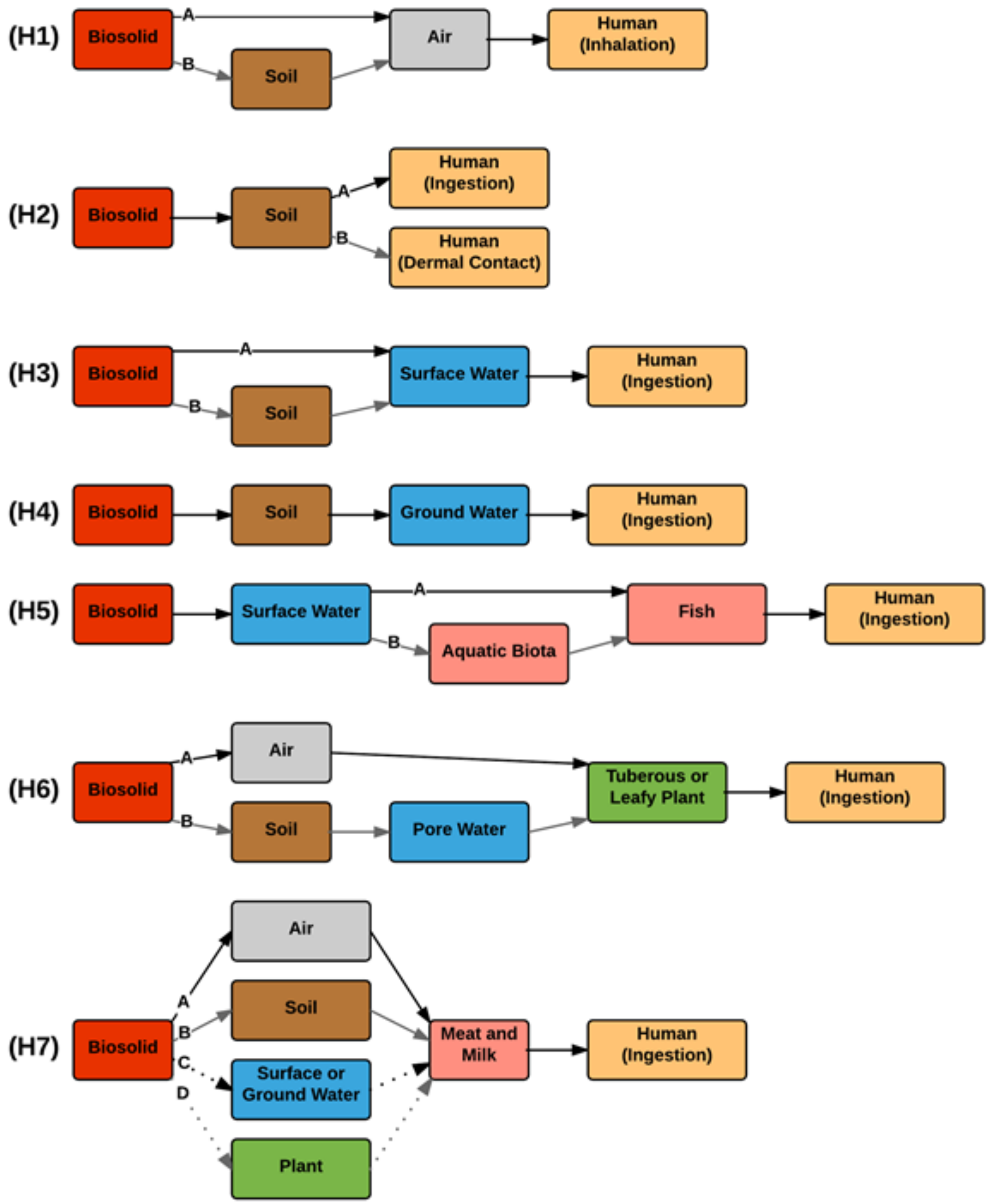

Figure 8: Generalized pathways for potential environmental exposure of humans to organic contaminants present in biosolids used in human health risk assessments. Risk assessments may consider different combinations of pathways depending on the specific system under study and on the scope of the risk assessment. Adapted from European Commission (2003)

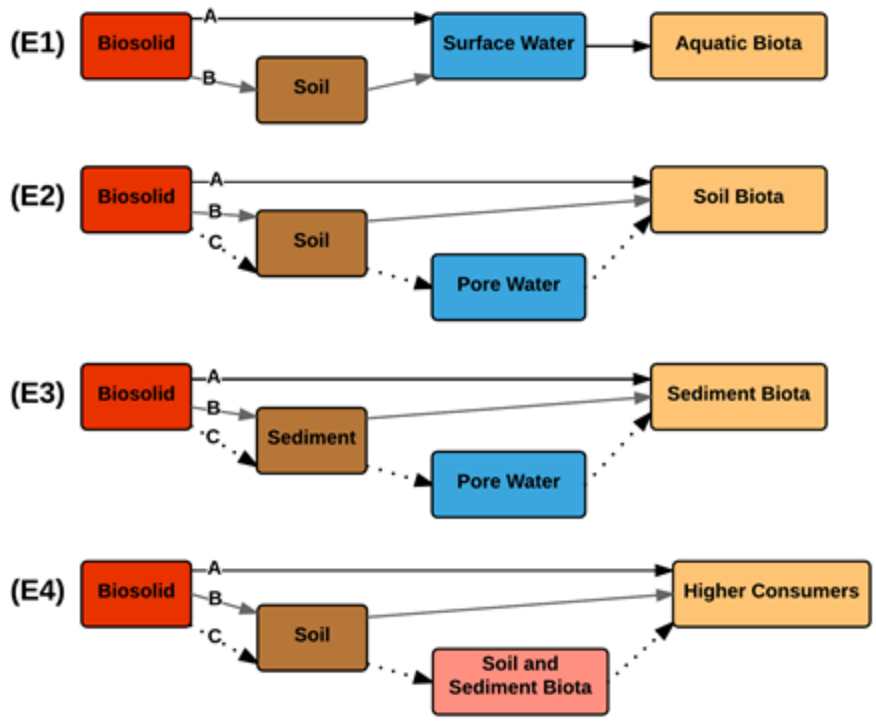

Figure 9: Generalized pathways for conducting environmental risk assessments on organic contaminants present in biosolids. If empirical data is not available, pathways can be modelled using mass balances and known transfer parameters such as distribution coefficients and bioaccumulation or biotransfer factors to predict exposure levels for comparison to a reference dose. Adapted from European Commission (2003)

The risk of a toxic chemical associated with certain exposure pathways will be affected by the persistence and bioaccumulation potential of that chemical (García-Santiago et al. 2016). For example, the amount of a contaminant present in soil is estimated to be greater for substances with a long half-life, and significant accumulation of persistent substances could occur with annual biosolids applications based on the percentage of the substance remaining on each subsequent application (García-Santiago et al. 2016). This would increase the PEC and consequently the risk for that substance. This is not an issue in jurisdictions, such as the Nicola Valley, where annual biosolids applications are reported to be uncommon. Likewise, the dose of a chemical delivered to a human through meat from cattle grazing on contaminated land (estimated through a mass balance calculation) depends on the dietary consumption of the substance by cattle and the biotransfer factor (García-Santiago et al. 2016). The log K_{ow} is a good predictor of the fat-adjusted biotransfer factor (USEPA 2005), thus lipophilic compounds will have a higher concentration in meat and will pose a greater risk to humans upon consumption than than the risk that direct soil exposure may have posed. It should be noted that presence is not equivalent to risk. Risk depends on exposure, which is function of bioavailability. The concentration that an affected population is exposed to is evaluated relative to concentrations that cause a deleterious effect in an affected population.

Summary of Human Health and Environmental Risk Assessments

A literature search was conducted with Google Scholar using the search term "quantitative risk assessment biosolids", and the titles of the first 1000 results were scanned to identify potential quantitative risk assessments conducted according to similar methodologies for which an index of risk was calculated for at least one of the pathways in Figs. 8 and 9. Only papers that considered risk from biosolids use were included. This was mainly treated sewage sludge, although the definition of biosolids and the degree of treatment required to qualify as "biosolids" are not consistent between all jurisdictions across the world. For this reason, some contaminant occurrence data from untreated WWTP solids is included. Risk assessments considering WWTP effluent impacts on aquatic organisms or pathways affecting human health were not included. In total, eight papers were identified that, in total, covered at least one exposure pathway in all of the major categories in Figs. 8 and 9, with the exception of H5 (Biosolids –> Surface Water –> Fish –> Humans) and E3 (Biosolids –> Sediment –> Sediment Biota) for at least one organic contaminant. However, comprehensive coverage of human and environmental risk assessment pathways was only available for a limited number of studies (García-Santiago et al. 2016; Snyder and O'Connor 2013). Human health pathways H1B, H2A, H2B, H6B, H7BCD, and total exposure were examined for carbamazepine, fluoxetine, triclosan, miconazole, and ciprofloxacin (García-Santiago et al. 2016). Human health pathways H1B, H2A, H3B, H4, H6B, H7B, H7D, and environmental health pathways E1B, E2B, E2C, E4A, E4B were examined for triclocarban (Snyder and O'Connor 2013). All of the studies were focused primarily on pharmaceuticals and personal care products.

The indices of risk presented in each paper were screened to identify risk pathways and organic contaminants for which there was a high risk (>1) or moderate risk (>0.1) to either human or environmental health. Risk pathways identified are presented in Table 11, and organic contaminants identified are listed in Table 12. Dietary exposure through contaminated plants was the only human health pathway identified as high risk, while aquatic biota, soil biota, and higher consumers were identified with a high environmental risk. The BC Organic Matter Recycling Regulations dictate that Class B biosolids applications must have 30 m distance from potable water sources or boundaries of properties zoned for residential or recreation, with guidance to increase the buffer distance if the biosolids application is on a slope. Biosolids are also not to be applied if the water table is closer than 1 m below the surface. Following these guidelines would reduce risk associated with direct transport to surface or ground water (H3A, H5, H7C, E1A). Human dietary exposure through meat and milk, and direct exposure of higher consumers to biosolids were identified as having a moderate risk. It should be noted that the risk is dependent on the specific substance considered and will vary depending on concentrations in biosolids and on the amounts applied to land. Biosolids application rates considered in the risk assessments varied between 5 to 50 Mg ha^{-1}.

Table 10: Summary of risk assessments included in this review, listing the hazards considered, the subject (human or environmental health), risk pathways, calculation of risk index, and a listing of the hazards or pathways for which a high or moderate risk was identified.

Ref.	Hazards Considered	Subject	Risk Pathways Considered	Assessment of Risk	Risk Quotient or Equivalent
(1)	TCS	Environ. Health	E2B: 4 Scenarios E4A and E4B: 6 Scenarios	Margin of Safety (95%) = Exposure Estimate / Toxicity Benchmark - Mean BS appl. rate = 19 Mg ha^{-1} - Probabilistic model of exposures run using Monte Carlo simulations	**>1:** Soil Microbes (Mixed Zone) **>0.1:** Soil Microbes (top 30 cm), Plants (Mixed zone), Short-tailed shrew, American robin, American kestrel
(2)	CBZ, FLX, TCS, MCZ, CPX	Human Health	H1B, H2A, H2B, H6B, H7BCD; Total exposure also considered	HQ (95%) = Exposure Dose / Reference Dose - BS appl. rate = 5 Mg ha^{-1} - Exposure estimated through mass balance models of chemical fate - Reference doses pulled from literature	**>1:** None **>0.1:** CBZ: H7BCD, Total; TCS: H6B, H7BCD, Total
(3)	7 Antibiotics, 7 EDCs, 16 Personal Care Products, 15 Pharmaceuticals	Environ. Health	E1B	HQ = Maximum Aqueous Concentration / Lowest Toxicity Value - BS appl. rate = 40 Mg ha^{-1} - Transport to pore water and surface water modelled to determine exposure - Lowest toxicity values reported in literature	**>1:** HHCB, AHTN, TCC, TCS, CPX, DXC, NFX, OFX, EYL, EOL **>0.1:** CBZ, FLX, 4NP, 4OP
(4)	15 PPCPs	Environ. Health	E2B	RQ = PEC / PNEC - BS appl. rate = 25 Mg ha^{-1} - PNEC from aquatic organisms by equilibrium partitioning method	**>1:** AZM, EE2, FRS, GFB, IBP **>0.1:** AHTN, CBZ, CIP, HHCB, MCZ, OFX, SMZ
(5)	18 Pharmaceuticals	Environ. Health	E2C	RQ = PEC / LOEC - BS appl. rate = 45 Mg ha^{-1} - PEC based on modelled porewater concentrations. - LOEC based on lowest available effects concentration for aquatic biota	**>1:** CFN, OFX, TET, TCC **>0.1:** TCS
(6)	ATL, CBZ, CPX, DPH, NPX, NFX, PNE, SBL, TNE, TRT, TCC, TCS	Human Health	H6B (Adult, Toddler)	HQ = EDI / ADI - BS appl. rate=3 to 22 Mg ha^{-1} - ADI based on lowest therapeutic dose with safety factor of 10^3 to 10^4, or NOEL with safety factor of 300. Values pulled from literature	**>1** Adult: CBZ, SLB, Total Toddler: CBZ, SLB, Total **>0.1** Toddler: TST, TCS
(7)	TCC	Human and Environ. Health	H1B (2 Scenarios), H2A, H3B (2 Scenarios), H4, H6B (2 Scenarios), H7B, H7D, E1B, E2B, E2C, E4A, E4B (2 Scenarios)	HQ = Exposure Dose (Concentration) / Reference Dose (Concentration) - BS appl. rate: Worst Case = 50 Mg ha^{-1}, 100 year = 5 Mg ha^{-1} $year^{-1}$ - Reference dose based on measured human, rodent, and soil organism toxicity	**Worst-case** **>1:** E4B: Shrew, American Woodcock E1B(2): Mysidopsis bahia **>0.1:** E1B(1): Osprey, River Otter, E1B(2): Ceriodaphnia sp., Pimephases promelas **100-year** **>0.1:** E2B: Eisenia fetida E4A: Shrew, American Woodcock
(8)	152 Pharmaceuticals 17 Personal Care Products	Environ. Health	E2B	RQ = (MEC or PEC) / PNEC -BS appl. rate = 5 Mg ha^{-1} -PNEC from aquatic organisms by equilibrium partitioning method	**Median value >1:** SFX, ENE, EOL, EYL, ACM, IBU, ROX, AZI, OFX, NPX, SAL, PPL **Median value > 0.1:** CMN, MPL, FLX

References: (1) Fuchsman et al. (2010); (2) García-Santiago et al. (2016); (3) Langdon et al. (2010); (4) McCarthy et al. (2015); (5) McClellan and Halden (2010); (6) Prosser and Sibley (2015a), Prosser and Sibley (2015b); (7) (Snyder and O'Connor 2013); (8) Verlicchi and Zambello (2015b).

Table 11: Risk assessment pathways for exposure to organic contaminants for which a high (>1) or moderate (>0.1) index of risk has been reported for at least one organic contaminant.

Index of Risk >1
H6B: Biosolid –> Soil –> Pore Water –> Tuberous or Leafy Plant –> Human
E1B: Biosolid –> Soil –> Surface Water –> Aquatic Biota
E2B: Biosolid –> Soil –> Soil Biota
E2C: Biosolid –> Soil –> Pore Water –> Soil Biota
E4B: Biosolid –> Soil –> Higher Consumers
Index of Risk >0.1
H7BCD: Biosolid –> Soil+Surface/Ground Water+Plant –> Meat and Milk –> Human
E4A: Biosolid –> Higher Consumers

Table 12: Organic contaminants for which a high (>1) or moderate (>0.1) index of risk has been reported for at least one risk pathway.

Index of Risk >1
Triclosan, Triclocarban, Sulfamethoxazole, Estrone, Estradiol, Ethynlestradiol, Acetominophen, Ibuprofen, Roxithromycin, Azithromycin, Ofloxacin, Naproxen, Salicylic acid, Propanolol, Furosemide, Gemfibrozil, Ibuprofen, Caffeine, Tetracycline, Carbamazepine, Salbutamol, Galaxolide (HHCB), Tonalide (AHTN), Ciprofloxacin, Doxycycline
Index of Risk >0.1
Clarithromycin, Metoprolol, Fluoxetine, Miconazole, 4-Nonylphenol, 4-t-Octylphenol

Organic Contaminants for Monitoring in Biosolids

In total 25 PPCPs were identified as high risk in at least one exposure pathway, and 6 were identified as moderate risk (Table 13). These compounds should be evaluated as candidates for monitoring, as there was a potential for harm identified that may be abated through controls on levels of specific contaminants in biosolids applied to land. Many of the substances identified through risk assessment were also detected at high frequency and at concentrations greater than $\mu g\ g^{-1}$ in biosolids from Salmon Arm, BC, included in the CCME survey of emerging substances of concern in biosolids (Hydromantis 2010) and listed in Table 13 (italicized compounds). These compounds are also good candidates for monitoring, as the high frequency of detection should maximize the information obtained with resources invested in sample analysis. Furthermore, since comprehensive human and environmental health risk assessments have generally not been completed for most substances, this will direct monitoring effort towards substances with the highest probability of exposure to an at-risk population through land application of biosolids.

Table 13. Most frequently detected substances in raw sludge and treated biosolids (Autothermal Thermophilic Aerobic Digestion) from Salmon Arm, BC, reported in the CCME survey of emerging substances of concern in biosolids (Hydromantis 2010). Italicized substances were also identified as having a risk from at least one of the pathways in Table 11 and are listed in Table 12.

	Detections		Median Concentration (ng/g dw)	
Substance	**Raw Sludge**	**Treated Biosolids**	**Raw Sludge**	**Treated Biosolids**
Triclocarban	3/3	3/3	3360	5010
Triclosan	3/3	3/3	8390	21500
Azithromycin	3/3	3/3	154	220
Caffeine	3/3	3/3	1270	4110
Carbamazepine	3/3	3/3	213	717
Ciprofloxacin	3/3	3/3	13000	6900
Clarithromycin	3/3	3/3	71	126
Diphenhydramine	3/3	3/3	451	612
Diltiazem	3/3	3/3	192	22
Enrofloxacin	3/3	0/3	14	NA
Erythromycin-H_2O	3/3	3/3	28	32
Fluoxetine	3/3	3/3	127	97
Miconazole	3/3	3/3	683	1350
Norfloxacin	3/3	1/3	410	154
Ofloxacin	3/3	3/3	326	245
Sulfamethoxazole	3/3	1/3	26	3
Thiabendazole	3/3	3/3	16	22
Trimethoprim	3/3	1/3	60	36
1,7-Dimethylxanthine	1/3	3/3	1030	1850
Bisphenol A	2/2	2/2	785	1220
DPMI	2/2	2/2	125	195
AHDI	2/2	2/2	565	370
HHCB	2/2	2/2	6975	8685
AHTN	2/2	2/2	3690	4440
ATII	2/2	2/2	760	1025
Musk Xylene	2/2	0/2	245	NA

The U.S. Environmental Protection Agency (USEPA) has devised a screening level predictive tool (PBT profiler) to help identify chemicals that potentially may persist, bioaccumulate, and be toxic to aquatic life (i.e., PBT chemicals). Demonstrated in Table 14 are the profiler's results for selected contaminants in sewage sludge showing persistence in soil (half-life in days), bioaccumulation (BCF), and chronic toxicity risk factors (fish). Clarke and Cummins (2015)

Table 14: Identification of risk factors persistence, bioaccumulation, and toxicity for selected contaminants from the U.S. Environmental Protection Agency Persistence Bioaccumulation and Toxicity (PBT) profiler (Clarke and Cummins 2015).

Chemical class	Persistence (half-life in soil [days])	Bioaccumulation (BCF aquatic organism)	Chronic toxicity (fish mg/L)
		POPs	
PCBs	360	25,000	0.005
PCDD/Fs	360*	14000*	0.0017*
Hepta BDE	360*	18,000*	Not estimated
HexaBDE	360*	4,800*	Not estimated
PentaBDE	360*	15,000*	0.00069
TetraBDE	360*	4,600*	0.003
PFOS	360*	56*	3
PeCB	360*	1200*	0.037*
		PPCPs	
Anti-convulsant			
Carbamazepine	75*	19	0.9*
Beta-blockers			
Atenolol	75*	3.2	1.1*
Propranolol	30	51	0.95*
Metoprolol	75*	4.5	5.3*
Sotalol	30	3.2	1.1*
Antimicrobials			
Triclosan	120*	640	0.071*
Triclocarban	120*	800	0.013*

*Indicates criteria have been exceeded (criteria are based on scientific principles and QSARs that have been used to screen for toxicity in the US Environmental Protection Agency's "New Chemical Program" for over 20 years). (Adapted from US Environmental Protection Agency 2013, see http://www.pbtprofiler.net/details.asp)

Knowledge Gaps and Opportunities for Improvement

Significant progress has been made in generating quantitative descriptions of organic contaminant persistence, bioaccumulation, and toxicity, to determine the risk of a substance to a susceptible population. However, progress is hindered by a lack of comprehensive empirical datasets for detailed risk assessment, and current approaches frequently depend on safety factors rather than a substantial body of supporting information (Malchi et al. 2015). Due to a lack of data, the mass balance exposure models are not always empirically validated. In the case of the equilibrium partition method used to establish soil toxicity thresholds based on aquatic toxicity, there can be significant variability in calculated values depending on the source of the data (McCarthy et al. 2015; Verlicchi and Zambello 2015). Some authors advocate strongly for the consideration of contaminant mixtures and metabolites in the assessment of risk, rather than individual substances, to account for the wide range of compounds detected in biosolids (Malchi et al. 2015). The risk assessment framework is in place, but requires additional work to produce complete datasets for assessing risk in the known biosolids contaminants.

Recommendation 3: That a quantitative risk assessment of land application of biosolids specific to the Nicola Valley be conducted. This risk assessment should consider pathways H1A, H6B, H7, E1B, E4 that either of concern based on studies reported in the literature or are poorly represented in the literature and are of particular concern to residents in the Nicola Valley. The risk assessment should include consideration of risks associated with exposure to candidate pharmaceuticals and personal care products (Table 13).

Recommendation 4: An organic contaminant monitoring program should be established for biosolids being applied in the Nicola Valley. This should include monitoring of concentrations in source materials and treated products.

7. Exposure Pathways to Biosolids and Composted Biosolids Land Applied in the Nicola Valley

In the previous section a review of quantitative risk assessments published in the literature was undertaken and considered in the context of the Nicola Valley. In this section we will consider the pathways of particular interest in the Nicola Valley in the context of the land application of biosolids and composted biosolids. In particular, we will focus on pathways involving direct injection by wildlife and direct inhalation of biosolids, bioaerosols and odors by humans.

In the Nicola Valley, concerns exist over the risks associated with land application of biosolids to native grassland and agricultural fields (Fig. 10). Risk to humans and the environment depends on the severity of, and the potential for exposure to, contaminants of concern. Potential exposure to volatile emissions, dust particles and bioaerosols occurs in proximity to biosolids storage or composting facilities (Brooks et al. 2005b; Dowd et al. 2000; Forcier 2002; Jenkins et al. 2007). Exposure can also occur during the transportation, unloading, application of the product and turning of biosolids or biosolids compost (Bhat et al. 2013; Forcier 2002). Plant uptake of contaminants, particularly heavy metals, can occur following application, whereas unincorporated biosolids also have the potential to adhere to the plant surfaces and enter the food chain (Blaine et al. 2013; Codling et al. 2014; Chaney et al. 1996; Gardner et al. 2012; Prosser et al. 2014; Speir et al. 2004; Wu et al 2015). Wildlife and livestock can consume contaminants directly from soil (E4B), or as a result of consumption of contaminated herbage (Chaney et al. 1996; Fries 1996; Hill et al. 2005; Washburn and Beiger 2011). The consumption of contaminated vegetation (H6), animal products (H7) and water (H3, H4) and the respiration of air-borne contaminants (H1) all serve as potential human exposure pathways.

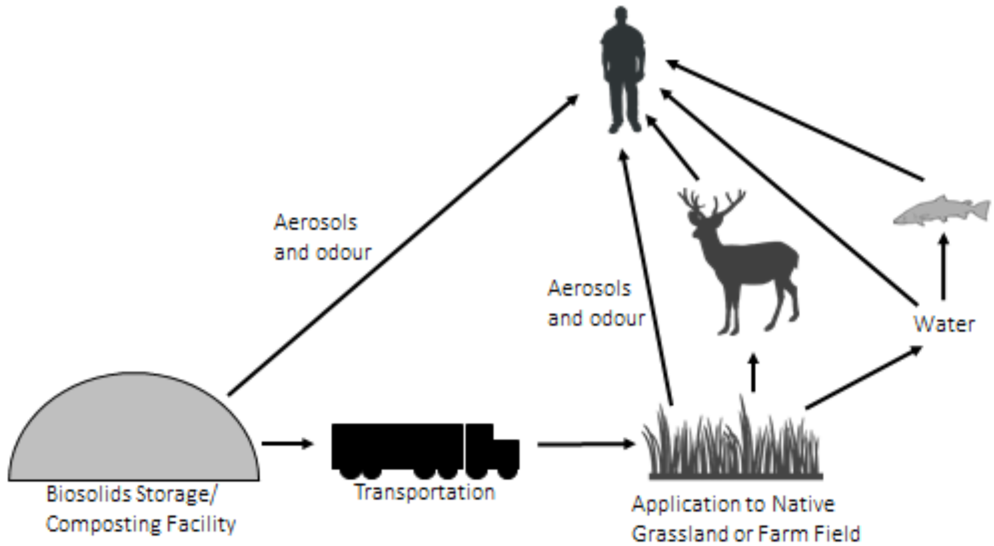

Figure 10: Selected potential human-exposure pathways to biosolids-derived contaminants applied in Nicola Valley, modified from (Chaney et al. 1996; Hill 2005) For a comprehensive list of generalized pathways, refer to Figure 8 and 9.

Biosolids → Soil → Plant (H6)

Uptake

Following land application of biosolids, contaminants may dissipate in the soil as a result of volatilization, degradation or other processes (Chen et al. 2014). Contaminants in soil pore water may be taken up into plant roots, a process driven by transpiration processes (Prosser et al. 2014; Wu et al. 2015), or by absorption to plant surfaces where volatized organics can be captured by lipophilic leaf surfaces (Chaney et al. 1996; Haynes et al. 2009; Topp et al. 1986). Evidence of uptake of biosolids-derived contaminants in agricultural-relevant plants includes: toxic metals or metalloids, such as Cd, Hg, Pb, As and Se, in a variety of edible crops including garden vegetables and fruits (Haynes et al. 2009); perfluoroalkyl acids (PFAAs) in greenhouse lettuce and tomato (Blaine et al. 2013); human pharmaceuticals in cabbage aerials, lettuce and other crops (Holling et al. 2012); and polycyclic aromatic hydrocarbon (PAH) and polybrominated diphenyl ether (PBDE) in tall fescue and wheat crops (Feng et al. 2014; Li et al. 2015). Once taken up by the plant, compounds can undergo phytodegradation or volatilization. Evidence suggests that many organic compounds, such as pyrene, benzene and trichloroethylene, can be degraded by plant enzymes in root and stem tissue (Haynes et al. 2009). Alternatively, persistent compounds and metals may be translocated and stored in stems or root structures, as dictated by partition coefficients and chemical characteristics of the contaminants (Prosser et al. 2014). Some species and tissues of plants have greater tendency to accumulate toxins than others. Carrot peels have a high affinity for lipophilic molecules and were shown to accumulate more than ten times more PCBs than sugarbeet (Chaney et al. 1996; Moza et al. 1979). After uptake into the plant, metal and organic contaminants, such as pharmaceutics, may or may not cause toxicity to the plants themselves (Haynes et al. 2009).

The presence of contaminants in biosolids-amended soil does not necessarily correspond to detectable uptake in plant tissue. Organic matter, aluminum and iron oxides, and other compounds in soils can bind metals, such as cadmium, and reduce their bioavailability (Gardner et al. 2012; Kukier et al. 2010). Overall, the mobility and bioavailability of metals in a given medium should be considered in determining potential environmental impacts (Alvarez et al. 2002; Banks et al. 2006; Cai et al. 2007; Haynes et al. 2009). In effort to describe the behaviour of metals in waste, laboratory fractionation schemes relate metals extracted under certain conditions to mobility and bioavailability in wastes and soil environments (Alvarenga et al. 2015). The scheme proposed by the European Community Bureau of Reference (BCR), and used extensively for biosolids characterization (Alvarez et al. 2002; Haynes et al. 2009; Wang et al. 2005), proposes a three-stage extraction with four fractions: (1), the easily available or exchangeable fraction in clay and organic matter phases; (2), the available, reducible fraction associated with Fe and Mn oxides associated with mineral matter unstable under anoxic conditions; (3), the relatively unavailable oxidizable fraction bound to organic matter or in cell walls; and (4), the residual fraction including metals in crystalline structure (Haynes et al. 2009; Hill 2005). A fifth fraction of carbonate-bound metals, susceptible to changes in pH, is also isolated in some studies and is considered bioavailable (Nomeda et al. 2008; Zorpas et al. 2008). Thus the fate of contaminants in biosolids or composted biosolids should consider the mineralogy, pH and redox status of the soils of the Nicola Valley environment in assessing bioavailability.

Metal Uptake

Following land application of biosolids or composted biosolids, the availability and mobility of metals is influenced by a variety of factors. The sorptive capacity and heavy metal content of the biosolids directly influences the availability of metals (Haynes et al. 2009). Other factors include physico-chemical properties of the soil or medium, environmental conditions, the chemical forms of the metal, target organisms and the ability of plant uptake (Amir et al. 2005; Haynes et al. 2009; Smith 2009b). If a high loading of heavy metals is observed in biosolids material, co-composting with an adsorbent material, such as zeolite, has the potential to decrease the fraction of available metals dramatically (Haynes et al. 2009). This practice may be required in some jurisdictions to dilute the metal and reduce its bioavailability. The repeated application of composted biosolids at rates studies (<30 Mg ha^{-1}) did not adversely affect semi-arid grassland in terms of metal content or fertility (Ippolito et al. 2010).

Despite elevated levels of heavy metals in their tissues, tolerant forage and cereal grain crops grown on biosolids amended soils tend to have increased growth and productivity and show no toxicity, even at high application rates (Codling et al. 2014; Gardner et al. 2012; Speir et al. 2004). This reflects, in part, the nutrient content of the biosolids. For reclamation of copper mine tailings in British Columbia, biosolids were applied at five rates from 50 to 250 dry Mg ha^{-1}, noting that 150 dry Mg ha^{-1} is the standard operational rate of mine reclamation in British Columbia (Gardner et al. 2012). A seed mix of wheatgrass, orchard grass, creeping red fescue, wild rye grass, alfalfa and alsike clover was planted and biosolids application resulted in significant increases in plant biomass relative to one-time fertilizer application or un-amended control plots. Composted biosolids applied annually for 4 years to a grazed ryegrass-clover pasture at rates of 0 to 200 dry Mg ha^{-1} resulted in enhanced fertility and productivity with no apparent adverse effects attributed to heavy metals in a field or growth room experiment with silver beet (Speir et al. 2004).

Decades after biosolids were applied to land, heavy metals and other persistent compounds were still available for plant uptake. For example, P and Zn were available for uptake in wheat 16 to 24 years after biosolids were applied (Codling 2014). Cumulative rates of biosolids application varied from 50 to 672 Mg ha^{-1} at selected sites (Codling 2014; Kukier 2010). Codling (2014) showed that wheat grown in biosolids-applied soil from Illinois contained 108 mg kg^{-1} of Zn compared to 48.2 mg kg^{-1} in non-amended soil. Two fold increases in Zn were also observed in wheat grown in soils from Minnesota and Maryland. Despite significant levels of metal uptake, the majority of treatments resulted in an increase in wheat biomass.

Organic Contaminant Uptake
At low rates of biosolids application, uptake of contaminants was not consistently above detection limits in several studies. Levels of PFAA in greenhouse conditions were predominantly below limit of quantification in lettuce and tomato grown in field soil amended with one application of biosolids (Blaine et al. 2013). Sabourin et al. (2012) applied 8 dry Mg ha^{-1} of dewatered biosolids to a small field experiment and found no consistent detection of 118 pharmaceuticals, 17 hormones and 6 parabens above the detection limit in triplicate treatments in four vegetable crops grown on the plots. Gottschall et al. (2012) detected no uptake in grain of wheat following application of 22 dry Mg ha^{-1} of dewatered municipal biosolids. In general, plant uptake of organic contaminants is low due to relatively low concentrations present and many transformations that occur in the soil environment (Haynes et al 2009).

Adherence
The adherence of biosolids-derived organic contaminants, metals and pathogens to plant tissue is of greater exposure risk to wildlife and human health than exposure following plant uptake (Chaney et al. 1996). When organic amendments, such as biosolids, are surface-applied to grassland or forage crops and are not incorporated into the soil, the residues can adhere to the aboveground plant canopy (Chaney et al. 1996). Chaney and Lloyd (1979) observed the adherence of biosolids to crops for prolonged periods after application, especially if biosolids were given the time to dry on the plants. Depending on moisture content, typically biosolids adhere to herbage at concentrations of 50 to 80 g sludge per kg herbage, both on a dry matter basis (Hill 2005). However, concentrations as high as 350 g sludge per kg herbage (DM) have also been observed (Hill 2005). Over time, the concentrations of biosolids on herbage decrease as residues are degraded or washed off plant surfaces (Chaney et al. 1996). The day after land application, Decker et al. (1980) found biosolids comprised of 22.3% of dry weight of forage and 18.6% of feces of grazing animals, on a dry weight basis (Chaney et al. 1996). However 21 days after application, the biosolids on forage was reduced to 5.39% (Decker et al. 1980). Washing herbage directly after application, via rainfall or spraying, can reduce the adherence of biosolids. Currently, many jurisdictions impose waiting periods between

biosolids application to standing forage and livestock grazing (Chaney et al. 1996; Haynes et al. 2009; Hill 2005). Waiting times of three weeks, six weeks, and one year have been implemented (Haynes et al. 2009; Hill 2005). Organic Matter Recycling Regulation (OMRR) for British Columbia restricts domestic animal grazing within the first 60 days after land application of Class B biosolids or compost with faecal coliform levels greater or equal to 1000 MPN per gram of total soils (BC 2002). However, no regulations are outlined for other Class B or any Class A managed organic matter. Also, there are no provisions in the regulations for undomesticated grazers (e.g. wild deer).

Incorporating or injecting biosolids into the soil can greatly minimize ingestion of biosolids-derived contaminants by livestock and wildlife and also serve to reducing odors associated with land application. Incorporation of land applied biosolids into the soil can also increase the degradation of organic contaminants (Al-Rajab et al. 2015). Following application of dewatered municipal biosolids on soil cores, dissipation of triclosan, triclocarban, and naproxen were significantly faster in subsurface aggregates than those on the surface (Al-Rajab et al. 2015).

Biosolids → Soil/Plant → Animals (E4)

"A major concern is transfer to and accumulation of heavy metals in the grazing animals" (Haynes et al. 2009). Grazers have immediate access to contaminated forage and soil when biosolids product is surface applied directly to herbage (Fries 1996; Hillman et al. 2003). This risk can be exacerbated when herbage is low because involuntary soil intake increases (Hillman et al. 2003). Soil intake can be as high as 30% for sheep, who graze close to the ground, when forage is sparse (Abrahams and Steigmajer 2003).

Livestock

In the literature, studies investigating the effect of biosolids diets on livestock typically use sheep and, to a lesser extent, cattle, (Chaney et al. 1996; Haynes et al. 2009; Hill 2005; Hill et al. 1998; Wilkinson et al. 2003). Stuczynski et al. (2007) applied biosolids at a rate of 300 dry Mg ha^{-1} with lime to Zn and Pb smelter wastelands. Although Pb and Cd levels found in hay (red fescue, tall fescue, sheep's fescue and Kentucky bluegrass mix) greatly exceeded current allowed thresholds, by 20- and 6- fold respectively, they did not affect the growth of calves. The concentrations of Pb and Cd in beef meat were below maximum permissible levels (MPL) in Poland, at 0.2 and 0.05 mg kg^{-1}, respectively. It is uncertain how the risks may be different for wildlife feeding directly on application site. It was concluded that lamb carcass meat was acceptable human food with low concentrations of Cd, Pb, and Cd in muscle and no signs of toxicity observed in grazing animals (Hill 2005). Sheep whose livers and kidneys exhibited levels of potentially toxic metals exceeding maximum permissible amounts had been offered extremely high concentrations of Pb, greater than 40 mg kg^{-1} DM intake, or exhibited high soil ingestion (Hill 2005). In a review of the literature, Hill (2005) also noted a potential transfer of Pb and Cd across placenta and in milk, but potential human health consequences were not determined in the studies reviewed.

Many organic compounds commonly found in biosolids, such as phthalates, acid phenols, and volatile aromatics, are readily metabolized and therefore, do not tend to accumulate in animal tissues (Fries 1996). However, persistent and lipophilic organic compounds, such as halogenated hydrocarbons, are a concern from land application of biosolids as they may accumulate in animal fat and tissues (Haynes et al. 2009). When fed 25 mg of DDT and DDE for 60 days, dairy cows excreted compounds in their milk fat (Fries et al. 1975, Fries et al. 1996). Accumulation of DDT was also observed in sheep through ingestion of soil (Harrison et al. 1970). Implementing waiting times between application and livestock grazing can significantly reduce risk of exposure (Haynes et al. 2009).

Wildlife

Land application of biosolids results in an increase in groundcover and augmentation in the nutritional value of plant life, thereby improving forage for wildlife consumption on grazing land that typically does not receive any inorganic fertilizers (Elfroymson et al. 1998; Pierce et al. 1998; USEPA 1991; Washburn and Begier 2011). Although potential exposure and toxicity pathways to wildlife have been identified in the literature, evidence is variable and site specific. In one study, biosolids application reduced Mo concentrations in mine tailings and plant tissue at a contaminated site (Gardner et al. 2012); however, concentrations remained above recommended ruminant health guidelines – an issue for grazing wildlife or livestock. However, it is important to consider that the original conditions of the site (mine tailings) would have been potentially toxic to grazers. Bean et al. (2014) observed changes in bird behavior after exposure to an antidepressant found in sewage sludge, but the results varied and the experiment was not performed in the field. Elfroymson et al. (1998) used a comprehensive review of field studies and mathematical risk models to examine wildlife risk from several ecosystems including Douglas-fir forests and semi-arid rangeland ecosystems. The researchers concluded that several small birds and mammals, including foxes and shrews, were not at risk from one time application of 40 Mg ha^{-1} of sewage sludge; however, there was potential for accumulation of Cu and Zn in white-tailed deer in eastern deciduous forest ecosystems (Elfroymson et al. 1998). In other risk assessment studies, Fuchsman et al. (2010) reported evaluated risk from triclosan using an exposure estimate and determined a low risk for short-tailed shrew, American robin and American kestrel; and Synder and O'Conner (2013) showed moderate toxicity for shrew, American woodcock and other microorganisms. However, minimal field data is available to valid these models.

In a Web of Science (all databases) search for "land appl*", "biosolids", and "wildlife", only 26 results were obtained. Several of these studies look at application of raw and treated sludge to forested land (Hegstrom and West 1989; Prescott et al. 2005; Thiel et al. 1989). In these systems, biosolids application tended to have a positive effect on soil fertility and yield (Prescott et al. 2005), but application of heavy metal rich products showed increased heavy metals in shrews, although no lesions were found on organs of these mammals (Hegstrom and West 1989). After application of biosolids to grass and/or shrub land, researchers have examined populations of meadow vole, and white-tailed deer (Anderson and Barrett 1982; Washburn and Begier 2011). There was no evidence of toxic effects of sludge on meadow vole population density, survival or reproduction in a field study on Oxford, Ohio where 1793 kg ha^{-1} of sludge was applied each month from May to September at N-fertilizing rates (Anderson and Barrett 1982). However, authors recommend long-term monitoring for potential chronic effects. In a field study of the long-term effects of biosolids application to North Carolina grasslands, it was determined that areas applied with biosolids had more bird visitation but no difference in visitation by white-tailed deer (Washburn and Begier 2011).

Overall, the potential risk to wildlife is difficult to accurately predict and there is a lack of field studies to validate mathematic findings. Studies are limited and of narrow scope; they do not study subtle changes in animal behavior and often do not consider exposure to cocktails of pharmaceuticals and other potential contaminants (Arnold et al. 2013; Brooks et al. 2012a; Backhaus and Karlsson 2014; Roccaco et al. 2014). Ecosystems are complex and a multitude of factors influence extent of exposure and risk including: the species of grazer, frequency of grazing, density of herbage, the amount and type of forage consumed, the potential supply of alternative fodder, the applied rate and chemical make-up of biosolids product, and precipitation, among others (Haynes et al. 2009; Hill 2005) There remains a number of unanswered questions that warrant further investigation. Hill (2005) discuss a number of these concerns including: a lack of evidence for the impact of potential toxic metals or organic contaminants on rumen microflora and bioavailability of these compounds in grazing ruminates; re-contamination of herbage with biosolids residue from rain splash on soil surface; and limited data on the adherence of organic contaminants on herbage.

Earthworms and Soil Fauna

There is a body of evidence on the chemical uptake and toxicity of biosolids application on soil micro- and macro- fauna (e.g. earthworms). Treated and untreated sewage sludge application resulted in elevated, and potentially toxic, metal and organic contaminant concentrations in earthworms and potential for magnification up the food chain (Beyer et al. 1982; Gaylor et al. 2013; Higgins et al. 2011; Kinney et al. 2008; Macherius et al. 2014; Na et al. 2011; Synder et al. 2011; Suthar et al. 2009). However, the relative increase in contaminant and toxicity depended on the concentration and availability of contaminants in the soil and in the applied sludge (Beyer et al. 1982; Na et al. 2011). Composted sludge also showed reduced toxicity to soil fauna (Andres and Domene 2005). Andres and Domene (2005) applied different sludge treatments to soil equivalent to 6% organic matter. The researchers observed negative effects following dry sludge application on several invertebrate communities and effects were still evident three years after sludge addition (Andres et al. 2011). However, the composted sludge treatment was able to effectively decrease ecological risks and reduce toxicity to Collembola compared to non-composted sludge (Andres and Domene 2005; Domeme et al. 2007). Although various anthropogenic waste indicators were observed in earthworm tissue as a result of biosolids application, indicators were also found in earthworms after application of swine manure at a comparable rate (Kinney et al. 2008). Banks et al. (2006) examined a variety of biosolids amended soils in the United States. Although some amendments decreased nematode survivability, no clear trend was observed. In this study, earthworms were negatively affected by some treatments, especially those with higher metal contaminants. Authors concluded that very few tests found that biosolids restricted survival, growth or reproduction (Banks et al. 2006). An additional study concluded low risk to invertebrates and plants from bisphenol A (BPA) following application of activated sludge biosolids (Staples et al. 2010). Overall, heavy metal and contaminant loading in biosolids should be considered prior to land application of biosolids to ensure that applied rates do not exceed potentially toxic thresholds to soil populations (Suthar et al. 2009).

Recommendation 5: Should direct ingestion by domestic and wildlife be found to be an exposure pathway of concern, the feasibility of the incorporation of land applied biosolids animals should be examined as a means of reducing direct exposure of biosolids to grazing.

Biosolids → Soil → Water (H3, E1)

The potential for transfer of organic and metal contaminants to surface and groundwater is dependent on a range of factors, such as the concentration and type of contaminants present, extent of microbial decomposition and volatization, and potential uptake by plants (Haynes et al. 2009). Hydrophilic, water-soluble organic contaminants are characterized by low organic carbon to water partitioning coefficients (K_{oc}) and are highly mobile in water (Haynes et al. 2009). If applied to land, these contaminants can also be transported lower in the soil profile and into the groundwater over time (Gottschall et al. 2013; McBride et al. 1997); posing significant environmental risk. However, water-soluble contaminants are frequently removed during the initial stages of sewage sludge and waste water treatment and therefore, are of minimal concern relating to land application of biosolids. In contrast, hydrophobic compounds generally remain sorbed to mineral and organic surfaces in soils and biosolids (Haynes et al. 2009). Metals and organic contaminants can become sequestered in the soil by binding to clay minerals, aluminum or ferric oxides or humic substances in the soil (Gardner et al. 2012). Although an increase in organic matter typically results in a decrease in the mobility of these compounds, there is the potential for movement of contaminants associated with dissolved Fe, Al, and organic matter (Lamy et al. 1993; McBride et al. 1997; Raber and Kögel-Knabner 1997).

The greatest risk for ground and surface water contamination occurs immediately after land application when soluble organic matter levels are elevated and when preferential flow of biosolids through macropores in soil is most probable or during significant rainfall events (Haynes et al. 2009). Because Nicola Valley is characterized by a relatively arid climate, it is expected that potential for runoff and leaching is low; however, heavy rainfall or snowmelt events increase the risk of surface runoff or leaching through coarse-textured soils. Concerns related to climate include: high groundwater table during late fall, winter and spring, and runoff concerns during snowmelt. These factors limit application and may also influence site access (SYLVIS 2008). Other important considerations include: soil permeability, soil depth, slope configuration and aspect. In the United States, a computer-based risk characterization screening tool (RCST) was developed to screen potential non-carcinogenic human risks associated with land application of biosolids and used to evaluate current regulatory limits associated with protecting groundwater (McFarland et al. 2012). Using the RCST model, when the depth to ground water at application site was maintained at 2 m, pollutant concentrations as large as 10 times the currently regulatory limit (US Code of Federal Regulations Part 503 – Ceiling Concentration Limits) could be applied safely at rates as high as 90 Mg ha^{-1} with no non-carcinogenic detrimental human health effects. In the OMRR, managed organic matter Class B biosolids and Class B compost with high levels of pathogens must be not applied to land where water table is within 1 metre of the surface (BC 2002). Also, these products cannot be applied within 30 metres of potable water sources. Extending these regulations to other products controlled under the OMRR should be considered.

Recommendation 6: Consideration should be given to extending the OMRR regulations pertaining to set back distances to water courses for the land application of Class B biosolids and Class B composts to other products controlled under the OMRR.

Several field studies have examined the effect of biosolids application on runoff and groundwater contaminants: at modest application rates, the runoff or leaching of metals appear to be insignificant (Gottschall et al. 2012; Hanief et al. 2015; Joshua et al. 1998; Rostagno and Sosebee 2001). In a study of surface-applied biosolids to rangelands, rainfall was stimulated on plots receiving different rates of biosolids application (0 to 90 Mg ha^{-1}) and different post-application ages (0.5 to 18 months) in the Chihuahuan Desert (Rostagno and Sosebee 2001). In the treatment receiving the highest rate of biosolids (90 Mg ha^{-1}) after 0.5 months, the surface run-off contained 4.96 and 97 mg L^{-1} of orthophosphate-P and ammonium-N, respectively. In this treatment, Cu also exceeded upper limit (0.50 mg L^{-1}) for livestock drinking water. However, contaminants were greatly reduced in lower application rates and longer post-application periods. In a similar study in Australia, biosolids (dewatered) were applied at rates from 0 to 120 dry Mg ha^{-1} and runoff and subsurface water data was collected over 1.5 years (Joshua et al. 1998). Overall, metals in the runoff were present in very low concentrations and not considered an environmental concern according to national standards. Although there was significant movement of Cu, Zn and nitrate moving down the soil profile in high rate treatments, at rates below 30 dry Mg ha^{-1} the movement of plant nutrient was low and almost negligible. In terms of organic contaminants, the United States EPA's Targeted National Sewage Sludge Survey (TNSSS) modeled movement of several sterols following biosolids land application (Chari et al. 2012). Although removal effectives of 10 different sterols was high (99%), toxicity modelling indicated that several hormones had significant leaching potential and could pose a threat to fathead minnows via run-off or leaching. However, this study models a "worst-case scenario" and may not adequately address degradation in soil or potential leaching or runoff based on depth to water table or distance to surface water. Avoiding application of biosolids in periods of high precipitation (or snowmelt) can greatly avoid environmental risks related to water contamination (Ma et al. 2015).

Biosolids → Soil → Water → Aquatic life

Several organic contaminants found in biosolids can also pose a health risk to aquatic organisms (Arnold et al. 2014; Jones et al. 2004). Even at very low levels, some PPCPs may exert negative effects on tadpoles, mussels and fish (Brodin et al. 2013; Hazelton et al. 2013; Schultz et al. 2011; Wu et al. 2015). Concentrations of estradiol as low as 1 ng L^{-1} induced vitellogen production, evidence of feminization, in male fish (Purdom et al.

1994) and dilute concentrations of oxazepam, a psychiatric drug, altered behaviour of wild fish (Brodin et al. 2013). However, although research suggests that low concentrations of estradiol may induce feminization, evidence in aerated soils typical of Canadian growing seasons suggest that estrogenic compounds are rapidly degraded (Lorenzen et al. 2006), therefore posing minimal risk of leaching and contamination of groundwater and aquatic systems. A majority of studies are performed in controlled, artificial environments and therefore, do not take complex processes of attenuation that occur in natural ecosystems into account. In a risk assessment by Snyder and O'Conner (2013) the *biosolids → soil → surface water → aquatic organisms* pathway was determined to have a high risk for triclocarbon (TCC) exposure to indicator aquatic organisms such as water fleas, fathead minnows and shrimp. However, when calculating the screening values to determine hazard, the model used the greatest measured TCC concentration recorded in United States surface waters. The authors inferred that this value likely represented contamination from raw sewage outflow rather than runoff from biosolids-amended field sites and therefore, may significantly over-estimate risk. In a modeling study in the EU, the non-steroidal anti-inflammatory mefenamic acid had the highest environmental impact on aquatic ecosystems. However, authors caution that the results present a particular geographic region and loading in biosolids that may not be relevant in other contexts (Morais et al. 2013).

Overall, there is very limited field-based research available that looks at the potential containment exposure to aquatic ecosystems following land application of biosolids in nearby grasslands or pastures. More research is needed to understand, first, the potential for land-applied biosolids to contaminate ground and surface-water in the Nicola Valley, and secondly, the potential impact on natural aquatic ecosystems with wild populations.

Biosolids → Air → Humans

Volatile emissions and odorants are a major public concern associated with treatment facilities and land application of biosolids materials and potential risk to human health (Barth et al. 2010; Burge and Marsh 1978; Forcier et al. 2002; Jenkins et al. 2007; McGinley 2002; Perez et al. 2006; Ziemba et al. 2013). Classes of odorant compounds associated with public concerns include sulfur compounds (hydrogen sulfide, mercaptans, and other sulfurous organics), nitrogenous compounds (ammonia, amines), and volatile fatty acids (Barth et al. 2010). Frequent reported complaints associated with odour exposure include irritation of eye, nose and throat, headache, nausea, diarrhea, heart palpitations, stress, drowsiness and alterations in mood (Schiffman et al. 2000). Symptoms typically appear at exposure and disappear shortly after, but may persist in sensitive individuals, such as asthmatics (Schiffman et al. 2000). Although reactions caused by odours are perceived as unpleasant by individuals, symptoms are not necessarily associated with a toxicological risk (Schiffman et al 2000). Rather, misperceptions of the risk associated with various biosolids-derived odours can significantly affect an individual's perception of the level of risk and their overall feeling of well-being (Andersson et al. 2013; Barth et al. 2010). Symptoms may be associated with annoyance, anxiety and frustration rather than physical irritation, as the concentration of many odorants in biosolids lies well below irritating thresholds (Cain et al. 2004). For example, at a sludge sludge treatment site, methyl mercaptan was found at a maximum concentration of 27 uL L^{-1}, whereas eye irritation occurs at concentrations greater than 575,440 uL L^{-1} (Cain et al. 2004). Often, humans are sensitive to concentrations of malaodors at magnitudes below toxic thresholds (Rosenfeld et al. 2007). Ammonia, a “fishy-smelling” nitrogenous compound, has an odour threshold concentration (OTC) of 26.6 μg m^{-3} while the recommended exposure limit for toxic effects is 18,000 μg m^{-3} (Rosenfeld et al. 2007).

Biological aerosols or bioaerosols are airborne particles adhering to dust and/or suspended in tiny droplets of water that originate from or consist of microorganisms, organic matter, plants, soil and other biota (Goyer 2001). Although bioaerosols are ubiquitous in the environment, they typically serve minimal risk to human health (Goyer 2001). However, abnormally high concentrations of bioaerosols can increase risk of allergic reactions, irritation or infection in humans following respiration (Goyer 2001). Fresh, wet organic materials, such as biosolids, provide the ideal conditions for presence and growth of microbiota and therefore, an increased risk for the release of bioaerosols. Potential human exposure to biological aerosols originating from biosolids can occur at storage or composting sites, and during turning, loading, transportation, unloading and application of raw or treated product (Forcier 2002). Using mathematical modeling to estimate the human-health risk of workplace exposure to microbial aerosols, Dowd et al. (2000) estimated a 3% risk of *Salmonella* infection for workers located 100 m downwind of biosolids with wind speed of 2 m s^{-1} and one hour of exposure. However, the authors caution that a ‘worst-case’ scenario was modeled and that additional epidemiological studies should be carried out to validate findings. Brooks et al. (2005) used a microbiological evaluation of 350 aerosol samples obtained downwind from a biosolids loading, unloading and application sites. Researchers concluded that the risk of microbial infection was low for residents near the sites. Furthermore, 16S ribosomal RNA sequences suggest that the majority of biological aerosols associated with biosolids land application in a dry, arid climate appeared to have been derived from the soil itself, rather than biosolids (Brooks et al. 2007).

A review of biosolids and bioaerosols in Québec concluded that human health risk from biosolids-derived biological aerosols was low for workers and nearby residents (Forcier 2002). In addition, risk to workers is further reduced by use of personal protective measures and adherence to guidelines restricting application rate and timing, storage, and also considering adequate buffer zones to surface water, ground water, and human residences (Forcier 2002). Currently, application of biosolids is British Columbia is under regulation outlined in the Organic Matter Recycling Regulation (BC 2002). Providing more inclusive waiting periods following application and restricting application in higher risk areas should be outlined for both Class A and Class B biosolids. Namely, avoiding areas with shallow water tables, nearby water courses or residential areas land can greatly diminish risk of water contamination. To ensure the safety of public and the environment, compliance to regulations may require enforcement by governing bodies.

Despite low risks cited in literature (Brooks et al. 2007; Cain et al. 2004; Dowd et al. 2000; Forcier 2002), there remains uncertainties in calculating overall risk to human health from bioaerosols and other volatile emissions derived from biosolids. The extent of biological aerosolization is dependent on a number of factors including: specific contaminants present in biosolids; the lifespan of airborne pathogenic organisms; weather conditions such as wind strength and direction, ambient temperature, humidity (Forcier 2002); and type of land application (Tanner et al. 2005). Minimal research has been done to examine the exposure limit thresholds of odorous chemicals in mixtures and to look at the public acceptability of blends of odours in mixtures (Cain et al. 2004), as well as the synergic and cumulative effect of biological aerosols on human health (Goyer 2001). Increasing public education with respect to biosolids and associated odorants and minimizing odour production in the management (Yang et al. 2003), storage and application of biosolids and biosolids products would improve public perception of biosolids recycling (Beecher et al. 2004).

Biosolids → Soil → Plant/Animal → Humans

Although humans can be exposed to biosolids-derived contaminants by consuming contaminated vegetation, animal products or water, the probably of exposure can be greatly reduced by implementing appropriate waiting periods and monitoring the content of heavy metals and persistent contaminants (Haynes et al. 2009).

Plant → Humans

Potentially dangerous levels of pathogens initially detected in applied biosolids can be reduced below toxic thresholds by implementing waiting periods between application and harvest (Brooks et al. 2012; Gale 2005). In one study, a four month delays between land application of biosolids and crop harvest, nearly all pathogen risks were reduced to below 10^{-4} (Brooks et al. 2012). Models suggest that risk to humans from vegetable and fruit consumption is low (Gale 2005; Hyland et al. 2015). However, metals, in particular Cd, Hg, Pb, As and Se, may accumulate in edible portions of crop plants and pose a threat to human health (Haynes et al. 2009; McLaughlin et al. 1999). Although the highest concentration of metals is typically found in the roots of plants, some metals have the tendency to accumulate in leafy tissues of particular plant species (Alloway and Jackson 1991; Haynes et al. 2009) and may be a high risk (Prosser and Sibling 2015). Therefore, monitoring of metal concentrations in vegetation grown in biosolids-amended soil is recommended (Haynes et al. 2009). Currently the OMRR restricts growth of food crops for human consumption with harvestable parts above ground for 18 months and that of below ground for 38 months for Class B biosolids with high pathogen levels (BC 2002). Regulations also restrict the application of Class A and B biosolids that will increase the level of metals beyond limits, depending on soil and site conditions (BC 2002).

Organic contaminants may accumulate in edible tissues of plants (Blaine et al. 2013; Chaney et al. 1996; Holling et al. 2012; Moza et al. 1979). However, at low rates of biosolids application, uptake of contaminants was not consistently above detection limits. Sabourin et al. (2012) applied 8 dry Mg ha^{-1} of dewatered biosolids to a small field experiment and found no consistent detection of 118 pharmaceuticals, 17 hormones and 6 parabens above the detection limit in triplicate treatments in four vegetable crops grown on the plots.

Animals → Humans
Metals can also accumulate in the tissues of grazing livestock and wildlife, typically concentrating in the liver, kidney, and to a lesser extent, muscle and fat of ruminant animals (Hillman et al. 2003). However, even when fed high biosolids diet, the retention of heavy metals is low. Johnson et al. (1981) fed steers a diet of 11.5% biosolids for 106 days. The resulting retention of metals was 0.09%, 0.06% and 0.30% for Cd, Hg and Pb, respectively, with no retention of Cu or Zn. Very low concentrations are typically found in carcass meat (Hill 2005) and based on reviewed literature, levels appear to remain below toxic thresholds for human consumption (Chaney et al. 1996; Hill 2005). Certain organic compounds, especially lipophilic, persistent, halogenated hydrocarbons, have a greater potential to accumulate in body fat or fat-containing products such as milk (Fries 1996; Haynes et al. 2009). However, concentrations of metals and organic contaminants in meat and milk and low and result in low risk for humans consuming these animals, a resulted echoed by risk assessment studies (Brooks et al. 2012; Chaney et al. 1996). Risk of exposure can be further lowed by providing additional forage for livestock can further reduce the risk of contaminated meat and animal products. Moving forward, researchers recommend more epidemiological studies and efforts to improve models of risk exposure and assessment to better understand the individual and cumulative risk of toxic metals, organic contaminants, and pathogens originating from land-application of biosolids on human and ecosystem health (Brooks et al. 2012; Dowd et al. 2000; Jenkins et al. 2007; Viau et al. 2011).

Biosolids vs. Manure: Highlighting the Relative Risk of Exposure
Review studies indicate that risk from bioaerosols, pathogens and contaminants derived from biosolids may be no more of a risk than those derived from animal sources. Several studies suggest that overall human health risks associated with exposure to manure-derived bioaerosols and pathogens can be comparable to or greater than those associated with municipal biosolids (Brooks et al. 2012; Forcier 2002):

- Significantly more manure is applied annually,
- Biosolids wastes are usually semi-solid and therefore, less susceptible to aerosolization;
- Manures typically contain levels of pathogens (*E. coli* and *Salmonella*) comparable to or greater than biosolids; and
- Restricted areas of application, especially for residential areas, are typically greater for biosolids than manure.

The bioavailability of metals and contaminants may also be higher in animal manures. Although biosolids contained a higher concentration of total estrogen, 923.9 ng g^{-1} dry solids (DS), than poultry manure at 286.7 ng g^{-1} DS, the poultry manure had a much higher desorption potential of 99.3 ng g^{-1} DS compared to potential of 3.9 ng g^{-1} DS for biosolids (Andaluri et al. 2012). The authors concluded that animal manures contribute to a significant amount of estrogen-type hormones in the environment. Raw manure sources can also provide a significant source of inorganic N and P that can result in environmental contamination following land application. Although nutrient run-off from biosolids application (Hanief et al. 2015) stimulated algal growth relative to reference soil in a laboratory experiment, land receiving equivalent amounts of inorganic N and P resulted in greater stimulation of eutrophication and N and P loading in aquatic systems.

Conclusions

1. Incorporation or injection of biosolids into the soil to reduce adherence to plant tissues is a best practice, reducing risk to wildlife and livestock, and increasing dissipation of organic contaminants (Al-Rajab et al. 2015; Chaney et al. 1996).
2. Adherence to the recommended waiting times, buffer zones and depth to water table restrictions in the OMRR following application of biosolids should continue to be enforced.
3. Implications of wildlife access to areas where biosolids have been applied should be studied.
4. There are limited comprehensive ecological studies that examine the effect of biosolids-application on natural land and aquatic ecosystems, including wildlife at all trophic levels.
5. There is a gap in the literature pertaining to the synergetic effect of mixtures of organic and metal contaminants and risk to humans and the environment (Arnold et al. 2013; Brooks et al. 2012a; Backhaus and Karlsson 2014; Roccaro et al. 2014).

8. Potential of Composting to Mitigate Risks Associated with Biosolids Application

The Composting Process

Composting is defined as a “managed process of bio-oxidation of a solid heterogeneous organic substrate including a thermophilic phase,” (CCME 2005). By facilitating the biological decomposition process, composting accelerates the degradation and stabilization of organic matter. To ensure the production of consistent and high quality compost products in Canada, while protecting human health and the environment, *CCME Guidelines for Compost Quality* (CCME 2005) were developed through the joint effort of the Canadian Council of Ministers of the Environment (CCME), the Bureau de normalization de Quebec (BNQ) and the Canadian Food Inspection Agency (CFIA).

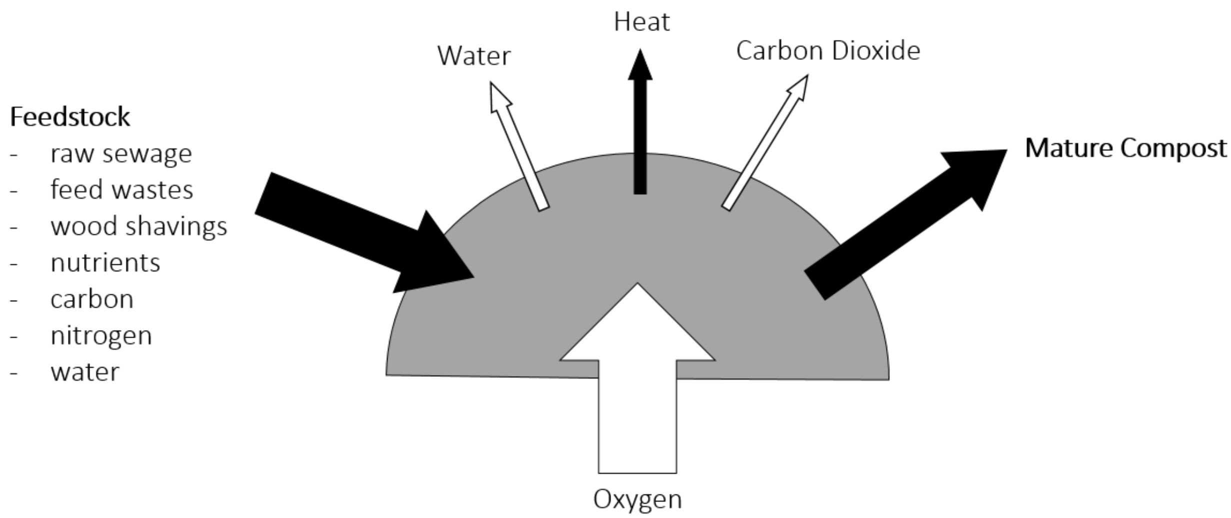

Figure 11: Simplified diagram of the composting process, adapted from (Rynk 1992)

The composting process, illustrated in Figure 11, begins with the selection and mixing of organic materials or feedstock, such as animal manures, shavings or municipal wastes. Sewage sludge is nitrogen-rich and often must be balanced with a carbon-rich bulking agent, such as straw or forestry by-products, to achieve a C:N ratio that favours microbial processes (Dumontet et al. 1999). With proper moisture content, aeration, and substrate availability, the microbial decomposers breakdown readily available sources of carbon. Under optimal conditions, these mesophilic organisms thrive and generate heat in the compost pile. Temperatures can reach and exceed 80^{o}C in the thermophilic or second phase of composting (Dumontet et al. 1999). Heat is important for the reduction of weed seeds and pathogens in the final compost product. Under CCME guidelines, all composts, other than those containing only yard waste, must achieve conditions of 55^{o}C for at least three days or longer, depending on method of composting used (CCME 2005). As microbial activity declines, temperatures decline and mesophilic decomposers re-colonize the maturing product. This cooling process characterized the third phase of the composting process (Fogarty and Tuovinen 1991). By this point, much of the accessible substrates have been decomposed. The final phase is the compost curing. To be considered mature, compost must be cured for at least 21 days and meet one of CCME indicators of low microbial activity (CCME 2005). By definition, when it is applied to plants, a *mature* compost should not have phytotoxic effects (CCME 2005).

To ensure high quality products, compost systems are designed to optimize the degradation process. Windrow systems consist of piling feedstock into long narrow piles typically between 9 to 20 ft in width and 3 to 12 ft in height to allow access to mix the compost (Rynk 1992). In actively aerated windrow systems, the pile is turned regularly to rebuild pore spaces in the material and allow air exchange. Turning the pile also releases trapped water vapour, carbon dioxide and other gases. Air exchange provides oxygen to support aerobic decomposition. In passively aerated windrows air can be supplied through perforated pipes embedded within or below the windrow, causing air to flow up through material (Rynk 1992). There are several variations on this system. In-vessel composting is an alternative to windrow composting and broadly describes a group of composting methods confined to a container or vessel (Rynk 1992). In-vessel systems rely on forced aeration and mechanical turning, resulting in an accelerated composting process.

In British Columbia, compost production and application is regulated under the Organic Matter Recycling Regulation (BC 2002). BC OMRR defines Class A and Class B compost classified according to standards including pathogen reduction processes, vector attraction reduction, quality criteria (including heavy metal concentrations), sampling and analysis protocols, and record keeping (BC 2002). For example, to produce Class A compost one of the following specific pathogen reduction processes must be used: either a windrow composting method involving periodic windrow mixing with a temperature not less than 55^0C last at least 15 days and no less than 5 turnings made during this high temperature period; or, a static aerated pile with mechanical aeration or enclosed vessel method with a temperature of not less than 55^0C maintained for 3 consecutive days (BC 2002). Class B compost has less stringent process requirements (40^0C or higher for 5 days with temperature of >55^0C for at least 4 hours during the period). Process requirements for vector attraction control and sampling frequency for pathogen reduction limits are outlined in the OMRR (BC 2002). Temperature and retention times must also be recorded during the production of Class A (not from yard waste) and B compost. Class A compost has higher quality standards with respect to pathogens and metals and its application is unrestricted. Class A must pathogen reduction limits (< 1,000 MPN g^{-1} dry weight (dw) fecal coliform), and adhere to sampling, analysis and record- keeping requirements. Biosolids may be used as a feedstock in the production of Class A compost provided that it does not exceed Class B biosolids quality criteria. Class A compost may be distributed without restriction. Class B compost must undergo pathogen reduction (< 2,000,000 MPN g^{-1} dw fecal coliform) and meet specified quality criteria.

Excessive odour production and leachate release into the environment are two major issues associated with composting sites. To address these issues, the OMRR requires that an environmental impact study and report be completed by a qualified professional prior to the design and operation of the composting facility (BC 2002). This impact report includes plans for odour reduction and leachate collection and treatment systems (BC 2002). In addition, specific regulations about the receiving, storage, processing and curing areas of the composting site must comply with certain facility requirements including: asphalt, concrete or impermeable surface to prevent release of leachate, roofing or covers to prevent runoff and reduce potential for leachate, and a leachate collection system (BC 2002). The regulation also provides limits for storage of residues (15 cubic metres in total), as well as the capacity of the facility for organic matter (BC 2002). Regulations specify that at least one half of the compost stockpiled at a composting site must be removed annually, beginning in the third year after start-up of the composting facility. Storage at the land application sites is also regulated, for example, buffer zones are required from watercourses for storage > 2 weeks (BC 2002).

Composting Biosolids: The Benefits

The composting process offers several benefits for managing organic waste. In general, composting reduces risks associated with raw organics in agricultural contexts, such as potential nitrogen deficiencies or excesses resulting in net immobilization or volatization of ammonia, respectively (Alvarenga et al. 2015). Composting stabilizes organic material, reduces the volume of waste and provides an opportunity for long-term storage of materials (Dumontet et al. 1999). Composting of biosolids addresses three public concerns: 1) a reduction in pathogens; (2) the degradation of organic contaminants and other emerging substance of concern (ESOC); and, (3) a reduction in the bioavailability of heavy metals.

Reduction in Pathogenicity

The composting process is an effective way to inactivate and destroy pathogenic organisms (Alvarenga et al. 2015). This sanitation is credited, in part, to generation of heat by microbes during the composting process, resulting in direct death or inactivation of pathogens during the thermophilic phase (Pereiraneto et al. 1986). A second explanation for the reduction in pathogens is the depletion of energy-rich, bioavailable substrates in finished compost. In the cooling and curing phases of the composting process, the readily available organic substrates have been degraded and the product is composed mainly of less-available recalcitrant forms of carbon. During this phase, the compost is recolonized by mesophilic organisms, including fungi and actinomycetes, that are often able to outcompete pathogens for the limited resources and may produce antibiotic compounds to discourage the growth of competitors (Dumontet et al. 1999). It should be noted that indicator pathogens are used as a relative measure of potential pathogenicity. Frequently studied indicators include: *Salmonella* spp., faecal coliform, and faecal streptococci, *Escherichia coli*, (Alvarenga et al. 2015; Dumontet et al. 1999; Pereiraneto et al. 1986). Although the selection of indicators has been disputed in other jurisdictions (Dumontet et al. 1999), *CCME Guidelines to Compost Quality* (CCME 2005) requires quantification of *Salmonella* spp. and feacal coliforms and the OMRR requires quantification of feacal coliforms (BC 2002). Requirements on the frequency of sampling and number of samples collected are outlined in the OMRR.

Composting of sludge typically results in a reduction of bacteria, viruses and parasites by at least 99.9% (Straub et al. 1993). Focusing on bacterial pathogens, Alvarenga et al. (2015) observed between 8.9 x 10^2 and 4.3 x 10^4 CFU *E. coli* g^{-1} of sewage sludge while levels were below detectable limits (< 1 x 10 CFU g^{-1}) in composted agricultural wastes and sewage sludge. In another study, composting sludge in an aerated static pile system was able to reduce *E.coli* and faecal streptococci from 10^7 to 10^2 organisms g^{-1} wet weight (Pereiraneto et al. 1986). Following static windrow forced aeration composting of sludge and wood chips, there was a reduction in faecal coliforms from 2.0 x 10^7 to 5.0 x 10^1 CFU g^{-1}. The same study observed more than a 1000-fold decrease in *Salmonella* spp., reducing levels below detectable limits (Dumontet et al. 1999). Furthermore, some studies suggests that the application of composted and partially composted biosolids to agricultural soils and in potting mixes may result in the additional suppression of selected soil borne plant diseases. In particular, immature sewage sludge composts were able to suppress *Pythium* damping-off in cucumber, a phenomena thought to be related to microbial activity (Kuter et al. 1988).

The risk associated with the pathogen content of composted biosolids can be compared to that of raw animal manures. In British Columbia, animal manures are land-applied annually, covering over 177,000 acres in the province in 2005 (STATSCAN 2006). In the United States, animal feeding operations are responsible for generating approximately 100 times the manure as sewage sludge (Gerba and Smith 2005). Several zoonotic pathogens can be found in manure including *E. coli* strain 0157, *Salmonella*, *Listeria* and *Campylobacter*, which can survive in stored slurries for several months (Nicholson et al. 2004). In a sampling of cattle feces from ten farms across the province of BC, 36% of calves were positive for the parasite *Giardia* (McAllister et al. 2005). In sampling nine commercial broiler chicken farms in Fraser Valley, BC, the average *E. coli* was 6.3×10^6 CFU g^{-1} litter sample (Furtula et al. 2010). Following land application of untreated manures, zoonotic pathogens in manure can survive in soil (Nicholson et al. 2004), and be transmitted to humans through the contamination of food crops or water (Gerba and Smith 2005). In contrast, according to CCME guidelines, composted human or animal waste must contain <1000 MPN g^{-1} total solids or no *Salmonella* at detection level of <3 MPN per 4g total solids, calculated on a dry weight basis (2005). Similar levels are required for Class A biosolids and compost in British Columbia (BC 2002). Thus pathogen content of composted biosolids represent far less of a risk than raw animal manure.

To achieve an adequate level of pathogen inactivation, the compost process must be effectively managed so that high temperatures (>55°C) are reached for a sufficient period of time (CCME 2005). Smaller pile size tended to contribute to lower pathogen levels (Brinton et al. 2009). The compost system and selected feedstock also have an effect on the extent of pathogen reduction and should be considered in process design (Dumontet et al. 1999). For example, a turned windrow composting system reduced faecal coliform levels in sludge sewage by less than 10-fold whereas a static windrow with forced aeration system decreased faecal coliform levels of the same feedstock more than 1000-fold (Dumontet et al. 1999). It is also suggested that in-vessel systems are a better option for pathogen reduction as they can more readily achieve uniform temperatures. The high temperatures achieved when co-composting sewage sludge with agricultural waste allowed for destruction of pathogens and parasites (Alvarenga et al. 2015). A final consideration is compost storage, as recolonization of composted waste by *Salmonella* spp. and other pathogens can occur, particularly in a product with high moisture content (Dumontet et al. 1999; Zaleski et al. 2005).

Degradation of Emerging Substances of Concern

In the treatment of industrial and domestic effluent, biosolids serve as a ‘sink’ for hydrophobic or non-water-soluble contaminants including pharmaceutical and personal care products (PPCPs), organic contaminants and other emerging substances of concern (ESOC). Composting of biosolids results in the reduction or complete elimination of some organic contaminants by accelerating the degradation of compounds through exposure to high microbial diversity and activity, high temperature, fluctuating pH, and changes in redox conditions due to shifting aerobic and anaerobic microenvironments (Xia et al. 2005). Due to microbial transformations during composting process and in soil systems, and the low initial concentrations in raw sewage, many of these compounds are considered of minimal risk (Haynes et al. 2009). Additionally, the bioavailability of these compounds has a major influence on how readily they are degraded by micro-organisms during the composting process.

Industrial Chemicals

A range of industrial chemicals have been quantified in fresh and composted biosolids including plasticizers, pesticides, and solvents (Hydromantis 2009). Alkylphenols and brominated flame retardants are also considered. The composting process reduced the concentration of a number of industrial toxins in biosolids including: a reduction in the chemical bis (2-ethyl-hexyl) phthalate (BEHP) of 64% (Gibson et al. 2007); reductions of 65-100% in nonylphenol (NP) concentrations following composting of biosolids in an investigation in the United States (Xia et al. 2005); reductions in di-ethylhexyl-phthalate (DEHP) by 91% and linear alkylbenzene sulfonates (LAS) by 99% (Moeller and Reeh 2003); and, evidence for degradation of various pesticides, polychlorinated biphenyls (PCBs), trinitrotoluene (TNT), and perchlorate (Hydromantis 2009). Limited evidence suggests that petroleum hydrocarbons can be reduced through composting process. Al-Daher et al. (1998) observed 46-59% degradation of total petroleum hydrocarbons depending on the bulking agent.

Brominated flame retardants, most commonly polybrominated diphenyl ethers (PBDEs), are shown to be relatively resistant to biodegradation and are able to persist in soil (Haynes et al. 2009). In an extreme example, after 20 years of annual application of Class B biosolids in southern Arizona, PBDE levels as high as 80 ng g^{-1} soil were detected for one congener (Quanrud et al. 2011). However, risk evaluation of exposure via inhalation, dermal exposure or ingestion determined that the health risk to humans of PBDEs was negligible. Additionally, it is important to note other sources of PBDEs in the soil, including air deposition which may be more relevant than risk from land applied biosolids. For instance, results in a study by Wu et al. (2007) suggest that indoor environment and diet play a role in human exposure to PBDE. These researchers observed strong, positive associations between PBDE in breast milk and house dust and consumption of dairy products and meat in a Massachusetts study.

Polycyclic Aromatic Hydrocarbons (PAHs)

The biodegradation of PAHs is variable: the removal of nine PAHs examined in a degradation study varied from 18 to 74% (Moeller and Reeh 2003). In a laboratory experiment examining the composting of biosolids and wood chips, phenanthrene persisted at 89 to 93% of initial concentration (Barker and Bryson 2002). However, in a composting experiment with preservative-treated wood and hog manure, PAH concentrations were reduced from 1000 mg kg^{-1} to 26 mg phenanthrene kg^{-1} and 83 mg pyrene kg^{-1}, which is equivalent to 2.6% and 8.3% of initial concentrations, respectively. It is suggested the longer composting times and the addition of manure to accelerate microbial activity aids in the decomposition of higher molecular weight PAHs (Barker and Bryson 2002).

Pharmaceuticals and Personal Care Products (PPCPs)

The composting process typically results in the reduction of biosolids PPCPs (Verlicchi and Zambello 2015a). Composting has been shown to reduce many pharmecuticals by >90% including ibuprofen, triclosan, and caffeine, Table 7 (Hydromantis 2010). Evidence suggests that the aerobic environment provided during the composting process is more favorable to degradation than anaerobic conditions for many compounds, including doxycycline (Hydromantis 2009). Some pharmaceuticals prove more resistant than others (Verlicchi and Zambello 2015a). Triclosan, a potential carcinogen and containment of public concern, has shown relative resistance to degradation in compost systems and has been recorded at median concentrations of 13,000 ng g^{-1} TS of biosolids, or 0.013% (Hydromantis 2009). Maximum values reported for the US were 133,000 ng g^{-1} dw, corresponding to percentage of 0.133% (Verlicchi and Zambello 2015a). Even at this concentration, this product still does not reach maximum regulated concentration in personal care products in Canada. Triclosan is currently used in a wealth of personal care products such as toothpaste, soaps, skin-care lotions and deodorants as an antimicrobial (Clarke and Smith 2011). According to Health Canada regulations, the maximum permitted concentration is 0.03% in mouthwashes 0.3% in cosmetic products (2015). Therefore, the highest reported values in biosolids contain a third of the concentration of triclosan as allowed in cosmetic products. This underscores the importance of the evaluation of the inclusion of these compounds in personal care products and the potential impact on waste streams.

Current data indicate that composting is an effective way to reduce industrial chemical contaminants in biosolids, but more data would strengthen this argument and address a wider range of chemicals (Hydromantis 2009). "More studies are needed to monitor the effects of composting on the degradation of other PPCPs in biosolids and determine the most effective composting treatment parameters," (Xia et al. 2005). Specifically, there is a lack of controlled studies which compare the raw or treated biosolids to a composted product and describe the impacts of these products on different soil systems (Briceno et al. 2007). An additional challenge is the ever expanding list of 'emerging' contaminants that could, and should, be tested.

Recommendation 7: The fate of major emerging substances of concern during composting should be documented.

Reduced Bioavailability of Heavy Metals

Anthropogenic wastes from industrial and household processes are often discharged in sewage. Two significant heavy metal contaminants of household effluents include Cu, from piping, and Zn, a popular component of household products such as deodorant, shampoo and aftershave (Comber and Gunn 1996; Haynes et al. 2009). In a study of Greater Vancouver Regional District biosolids, metals of concern included Cu, Cr, Sn, Cd, Zn, Ag, Mo, Se, Ni, Ba, and Pb, in order of greatest potential concern (Bright and Healey 2003). In contrast to organic contaminants, heavy metals are non-biodegradable; therefore, are resistant to the composting process and can persist in soil environments following land application (Haynes et al. 2009). Furthermore, the composting process may serve to concentrate these elements. In a study of six sewage sludge composts, Cd increased by 12-60%, Cu by 8-17%, Pb by 15-43% and Zn by 14-44% compared to raw sludge (Cai et al. 2007). This increase is attributed to the mineralization and volatilization of CO_2 and other decomposition products formed during the composting process (Cai et al. 2007; Smith 2009b). Alternatively, decreases in heavy metal concentrations following the composting of biosolids is also observed as a result of leaching of heavy metals from the compost pile (Cai et al. 2007) or a dilution effect resulting from mixing raw sludge with bulking agents with low heavy metal loading or mixing with soil following land application.

An important consideration is that the total concentration of metals in compost may not provide an accurate estimation of potential eco-toxicity. As a result of the increased affinity of metals for binding with composted organics, literature suggests that composting may result in a decrease in the availability of heavy metals in biosolids (Haynes et al. 2009; Smith 2009b). Heterogeneous organic and humic substances have a large number of reactive sites that are able to coordinate metal cations and other forms through a range of forces of attractions including chelation, adsorption, complex formation and co-precipitation (Haynes et al. 2009). Metals may also bind to other metals and inorganic compounds in the soil, as Kukier et al. (2010) suggests the involvement of soil organic carbon, iron oxides and phosphorous compounds in the reduction of Cd phytoavailabily in long-term biosolids amended soil in Illinois. The composting of biosolids and sewage sludge frequently increases metal content in oxidizable or residual fractions (Alvarez et al. 2002). Although composted sludge was characterized by higher concentrations of metals than digested sludge for several elements, following fractionation, a greater proportion of Cd, Cu, Mo, and Ni were observed in unavailable residual fraction of the composted product (Alvarez et al. 2002). In a study looking at changes in metal fractions during the composting of sewage sludge, bioavailable fractions of Cu, Zn and Pb tended to decrease over time whereas Ni was more variable (Amir et al. 2005). In another study, the composting process resulted in lower exchangeable and greater residual fractions of Cr, Cu, Fe, Mn, Ni, Pb, and Zn (Zorpas et al. 2008). The transformation of raw organic matter to stable humic substances during composting can allow the conversion of some metals into stable organic forms (Amir et al. 2005). However, not all studies report lower availability. Although Nomeda et al. (2008) observed a decrease in available Cu following composting, they report an increase in available Zn, Mn and Pb.

Due to their persistence in soil systems, heavy metals are a concern in the long term. Elevated levels of metal contaminants can be found in soils receiving biosolids decades following application. In an study of wheat crop in Maryland, Minnesota and Illinois soils, elevated levels of phytoavailable metals were found in soils 16 to 24 years following biosolids application, regardless of soil or biosolids source (Codling 2014). Cumulative rates of biosolids application varied from 50 to 672 Mg ha^{-1} at selected sites (Codling 2014; Kukier 2010). Ultimately, more research must be done to clearly understanding the short and long-term effects of these contaminants on these dynamic and variable soil systems.

Conclusions

1. The limited evidence that is available, suggests that composting is an effective treatment of biosolids in terms of reducing potential risk to humans and the environment (Alvarenga et al. 2015; CCME 2005; Haynes et al. 2009).
2. Controlling and monitoring temperatures during compost process is critical for pathogen and contaminant reduction (Alvarenga et al. 2015; CCME 2005; Dumontet et al. 1999).
3. Adequate storage conditions and monitoring of finished compost can minimize risk of re-colonization of pathogens (Zaleski 2005).
4. To adequately evaluate the potential eco-toxicity of heavy metals and ESOC in biosolids and composted biosolids, the bioavailability of elements and compounds should be considered in addition to their respective concentration (Haynes et al. 2009; Smith 2009b).
5. Reducing contaminants in composted biosolids begins with a reduction in load at source. Past enforcement of regulations regarding maximum metal loading by municipalities has resulted in decrease in heavy metal concentrations of Cd, Cr, Pb, Ni in Canada and United States in 1980s and 1990s (Haynes et al. 2009).
6. More comprehensive research is required to validate and understand the relative risk of biosolids compared to other organic wastes commonly applied to agricultural land across the province.

9. *Potential of thermal treatment of biosolids to mitigate risks associated with biosolids application*

Thermal treatment of sewage sludge offers a means to reduce potential toxicity and to generate energy and stable soil amendments (Rulkens 2008; Egan 2013). The net environmental benefit of thermal treatment can be further enhanced when biosolids are used as an energy source, replacing fossil fuels and ensuring sufficiently high temperatures to result in the destruction of most organic contaminants. This becomes particularly attractive as carbon markets develop in response to climate change mitigation efforts. Whether the process is an oxygen-limited process designed to generate a fuel source or a co-generation application, the opportunities to reduce the environmental impact of sewage sludge is an attractive but unproven new opportunity. The processes necessary to ensure and document effective energy recovery and the complete destruction of toxic organics are still being developed and optimized. Here we will review the early stage results on the range of thermal treatment options.

The simplest form of treatment is combustion where energy is released from the biosolids in the presence of oxygen resulting in the release of energy, carbon dioxide and the production of ash. In combustion there is no recoverable energy retained in the gaseous products. In torrefaction, pyrolysis and gasification reduced oxygen availability and elevated temperatures transforms the waste product to produce liquid and gaseous energy products as well as biochar. Torrefaction is essentially a pre-treatment process where heating is used to improve the thermal properties of the sewage sludge for subsequent energy generation. Egan (2013) contends that the most viable biosolids management strategy is energy generation with nutrient recovery and beneficial use of the by-product.

The application of biosolids-derived biochar to soil in place of biosolids has potential to minimize organic micro-constituents discharged to the environment (Ross et al. 2016)

Thermal Treatment Terminology

Incineration *is a waste treatment process that involves the combustion of organic substances contained in waste materials. Incineration and other high-temperature waste treatment systems are described as "thermal treatment". Incineration of waste materials converts the waste into ash, flue gas, and heat. The ash is mostly formed by the inorganic constituents of the waste, and may take the form of solid lumps or particulates carried by the flue gas. The flue gases must be cleaned of gaseous and particulate pollutants before they are dispersed into the atmosphere. In some cases, the heat generated by incineration can be used to generate electric power. Incineration is also referred to as "complete combustion" (Egan 2013)*

Combustion is a high-temperature exothermic redox chemical reaction between a fuel and an oxidant, usually atmospheric oxygen, that produces oxidized, often gaseous products, in a mixture termed as smoke. Combustion in a fire produces a flame, and in some materials, the heat produced can make combustion self-sustaining in others an additional energy source is required. The product gas of combustion does not have any useful heating value.

Torrefaction is a mild form of pyrolysis at temperatures typically between 200 and 320 °C. Torrefaction changes biomass properties to provide a much better fuel quality for combustion and gasification applications. Torrefaction is often used to process biosolids in advance of pyrolysis.

Pyrolysis is a thermochemical decomposition of organic material at elevated temperatures in the absence of oxygen (or any halogen). It involves the simultaneous change of chemical composition and physical phase, and is irreversible.

Gasification is a process that converts organic or fossil fuel based carbonaceous materials into gaseous fuels or chemicals. This is achieved by reacting the material at high temperatures (>700 °C), without combustion, with a controlled amount of oxygen and/or steam. The resulting gas mixture is called syngas (from synthesis gas or synthetic gas) or producer gas and is itself a fuel. The power derived from gasification and combustion of the resultant gas is considered to be a source of renewable energy if the gasified compounds were obtained from biomass.

from Wikipedia

Thermal treatment can be done on biosolids alone or in combination with other energy sources. There has been considerable work on the optimization of these processes to increase thermal yield. Egan (2012) states "Co-combustion results in the thermal destruction of toxic organics in the sludge (Otero et al. 2002)"; however, Otero et al. (2002) report on the thermal characteristics of the process and do not document the destruction of organic contaminants. Otero et al. (2002) states that "sludge combustion enjoys a combination of several advantages that are not found in other treatment alternatives, including a large reduction of sludge volume to a small stabilized ash, which accounts for only 10% of the volume of mechanically dewatered sludge, and thermal destruction of toxic organic constituents" citing Vesilind and Ramsey (1996). Vesilind and Ramsey (1996) merely report on the change in heating value as a result of processing and do not perform any direct measure of the reduction in toxic organics. Similarly, Egan (2012) states "metals in the resulting ash are more stable than metals in sludge prior to combustion (Otero et al. 2002)" but again, Otero provides no direct evidence an attributes the statement "combustion of sludge ensures a higher stability of heavy metals in the ashes as compared to the parent sludge" to Albertson et al. (1992) a proceedings paper that we were not able to access. Atienza-Martinez et al. 2013 found that torrefaction increased the energy density of the sewage sludge prior to pyrolysis. Poudel et al. 2015 found optimum temperature for the torrefaction of sewage sludge in terms of its energy and mass yield was 300-350 degrees C, however they did not examine the toxicity of the resulting solid material. It is critical that the ability of these treatment processes to reduce toxic loading through out the the life cycle (air, water, soil and biota) be clearly documented before they can be considered as viable alternatives.

Polyaromatic Hydrocarbons
The effect of thermal treatment on the content of polyaromatic compounds is variable. In some cases, treatment has been shown to decrease of PAH content (Zielinska and Oleszczuk 2015b). In other cases, pyrolysis has also been shown to result in the formation of PAHs (Liu et al. 2008), polychlorinated dibenzo-p-dioxins (Weber and Sakurai 2001) and polychlorinated dibenzofurans (Weber and Sakurai 2001). The pyrolysis of sewage sludge has been shown to result in a decrease of PCDD/F (Bayer and Kutubuddin 1994 as cited by Weber and Sakurai 2001). The extent of PAH formation during pyrolysis of sewage sludge is influenced by particle size (Dai et al. 2015). The pyrolysis of sewage sludge to biochar mobilizes some PAHs out of the sewage sludge and the net effect is a reduction of their concentration in biochars (Zielinska and Oleszczuk 2015b). The conversion of sewage sludge to biochar significantly reduced the content of PAHs from 8- to 25-fold depending on pyrolysis temperature and kind of sludge (Zielinska and Oleszczuk 2015b). The concentration of the most hazardous PAHs (5- and 6-ring) in sewage sludge-derived biochars was much lower compared to source material (Zielinska and Oleszczuk 2015b). The pyrolysis of sewage sludge resulted in a significant reduction in toxicity towards of the sludge based on test organisms (Zielinska and Oleszczuk 2015a).

Metals
Shao et al. (2015) report that the environmental risk of Cu and Zn were significantly reduced in biochar and the risk level of Cr was slightly reduced after pyrolysis or liquefaction. Yuan et al. (2015) found the toxicity of Pb, Zn, Ni, Cd, As, Cu and Cr in the biochars were lower than that in the sewage sludge although the pyrolysis process increased the concentration of the heavy metals in the biochars. The lower toxicity of these metals despite increased concentration reflects the lower bioavailability of the metals in the biochar. Zielinska et al. (2015) reported an increase in trace metal content (Pb, Cd, Zn, Cu, Ni and Cr) as a result of pyrolysis, but reduced bioavailability of the

trace nutrient elements (Mn, Fe, Zn and Cu). It should be noted that these toxicity assessments are generally made under laboratory conditions or on model systems and are seldom based on land applied biosolids. The assessment of metal bioavailability is much more difficult in these situations as the soil often attenuates metal availability.

Comparison of the concentration of metals, metalloids and PAHs with background soil concentrations, concentrations applied to the regulation of composted materials and European Union (EU) regulations relating to the application of sewage sludge to agricultural land suggest low risk associated with the concentrations of potentially toxic elements observed in biochars (Freddo et al. 2012). Collectively, these results suggest that environmental impacts attributable to metals, metalloids and PAHs associated with biochar following its application to soil are likely to be minimal (Freddo et al. 2012).

Emerging Substance of Concern

There has a limited amount of work on the impact of thermal treatment on emerging substances of concern. Those studies that have been performed generally demonstrate the destruction of these compounds during thermal treatment. Pyrolysis of biosolids demonstrated triclocarban and triclosan removal (to below quantification limit) at 200 °C and 300 °C, respectively (Ross et al. 2016). Substantial removal (>90%) of nonylphenol was achieved at 300 °C as well, but 600 °C was required to remove nonylphenol to below the quantification limit (Ross et al. 2016).

While the optimization and demonstration of thermal treatments is in its infancy, early studies suggest that there is promise in the thermal technologies to recover energy and reduce the toxicity of sewage sludge and the biosolids and biochar produced from this material. The practicality and operational performance of the particular approach taken is a subject that requires more detailed study involving local considerations.

10. Recommendations for monitoring of biosolids

Risk perception can often exceed actual risk as a result of dread based on a lack of knowledge. One of the important approaches to managing risk perception is providing the community with information as to the composition of the materials being applied and its compliance with accepted guidelines. It does not appear that the composition of the biosolids being applied to the Nicola Valley are routinely being communicated to that community. We would recommend that this be done in the future.

A more detailed monitoring of organic contaminants to deal with uncertainties as to the composition of the waste stream and its reduction as a result of treatment should be considered. Based on the most frequently detected compounds in biosolids from BC (Hydromantis 2010), and the compounds identified as a risk through various risk assessments reviewed for this report, the following compounds should be given a high priority for monitoring: Triclocarban, Triclosan, Azithromycin, Caffeine, Carbamazepine, Ciprofloxacin, Clarithromycin, Fluoxetine, Miconazole, Ofloxacin, Sulfamethoxazole, Galaxolide, Tonalide.

11. Recommendations for understanding and reducing risk

Recommendations Regarding Risk Management

Risk reduction is the first step in risk management. By reducing the addition of toxic compounds to our waste stream through source reduction initiatives and the management/recovery of toxic components we reduce their concentration in the waste stream. Initiatives such as the Medications Return Program recover pharmaceuticals. Efforts to eliminating or reducing the 'unnecessary' compounds, including some of the emerging substances of concern (e.g., triclosan) and persistent metals from consumer products;

Second is to manage the various exposure pathways present:

1. Use of personal protective equipment for workers (Forcier 2002) and implementation of transportation, loading and land-application guidelines to minimize release of bioaerosols; including implementation of the OMRR's buffer zones and restriction of application requirements during times of heavy rainfall and snowmelt to avoid surface or ground water contamination (Haynes et al. 2009);

2. Consider incorporation or injection of biosolids into the soil to reduce adherence to plant tissues and therefore, reduce risk to wildlife and livestock (Chaney et al. 1996);

3. The implementation of waiting times between application of biosolids and livestock exposure (Haynes et al. 2009; Hill 2005), supplying alternative forage crops and rotating livestock to non-biosolids applied pastures, and (if applied to native grassland) deterring wildlife grazing during critical waiting times after application;

4. Development of epidemiological studies of potential short- and long-term effects to consumers and the environment (Hogue et al. 1984), especially persistent metals and organics (Codling 2014);

5. Monitoring of metal and persistent organic compounds in edible crops grown in soil receiving biosolids application (Haynes et al. 2009).

The processing of sewage sludge or the biosolids produced from it can significantly reduce risk. Overall evidence suggests that composting is an effective treatment of biosolids in terms of reducing potential risk to humans and the environment (Alvarenga et al. 2015; CCME 2005; Haynes et al. 2009). Whether the risk is reduced sufficiently to be "acceptable" is a political decision involving a more detailed risk assessment based on the materials and circumstances found in the Nicola Valley, documented effectiveness of treatment processes and consultation with the residents of the Nicola Valley.

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13. Glossary

K_{ow}	Octanol-Water Partition Coefficient
K_d	Solids-Water Distribution Coefficient
K_{oc}	Organic Carbon-Water Distribution Coefficient
pK_a	Logarithmic Acid Disassociation Constant
ADI	Acceptable Daily Intake
BCF	Bioconcentration Factor
BMF	Biomagnification Factor
BTF	Biotransfer Factor
EC_{10}	10% Maximal Effective Concentration
EC_{50}	50% Maximal Effective Concentration
EDI	Estimated Daily Intake
HQ	Hazard Quotient
LOEC	Lowest Observable Effective Concentration
MEC	Measured Environmental Concentration
NOEL	No Observable Effect Level
NOEC	No Observable Effect Concentration
PPCPs	Pharmaceuticals and Personal Care Products
PEC	Predicted Environmental Concentration
PNEC	Predicted No Effect Concentration
QSARs	Quantitative Structure-Activity Relationships
RQ	Risk Quotient

Chemical Abbreviations

4NP	4-nonylphenol
4OP	4-t-octylphenol
ACM	Acetominophen
AHTN	Tonalide
ATL	Atenolol
AZI	Azithromycin
CBZ	Carbamazepine
CFN	Caffeine
CMN	Clarithromycin
CPX	Ciprofloxacin
DPH	Diphenhydramine
DXC	Doxycycline
ENE	Estrone E1
EOL	17β-Estradiol E2
EYL	17α-Ethynlestradiol EE2
FLX	Fluoxetine
HHCB	Galaxolide
IBU	Ibuprofen
MCZ	Miconazole

MPL	Metoprolol
NPX	Naproxen
NFX	Norfloxacin
OFX	Ofloxacin
PFAA	Perfluoroalkyl acids
PNE	Progesterone
PPL	Propanolol
ROX	Roxithromycin
SAL	Salicylic acid
SBL	Salbutamol
SFX	Sulfamethoxazole
TCC	Triclocarban
TCS	Triclosan
TET	Tetracycline
TNE	Testosterone
TRT	Triamterene

Class A Biosolids – Under the BC OMRR regulations "Class A biosolids contain lower fecal coliform densities (< 1,000 most probable number (MPN) g^{-1}) and lower trace element concentrations than Class B biosolids. Achieving stringent quality standards allows for more liberal distribution and use of Class A biosolids under the OMRR. The criteria for Class A biosolids are provided in Table 1. Refer to Section 3.4 for additional information on land application and distribution requirements."

Class B Biosolids - Under the BC OMRR regulations "Class B biosolids are subject to less stringent trace element and fecal coliform requirements (< 2,000,000 MPN g^{-1}) than Class A biosolids. As such, they are subject to more land application and distribution restrictions. The criteria for Class B biosolids are provided in Table 1. Refer to Section 3.4 for additional information on land application and distribution requirements."