

Summary, Evaluation and Integration of Atmospheric Deposition Monitoring in the Athabasca Oil Sands Region



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Summary, Evaluation and Integration of Atmospheric Deposition Monitoring in the Athabasca Oil Sands Region

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This report has been reviewed by workshop participants as listed in Appendix D.

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Table Of Contents

Foreward.....	1
1.0 Executive Summary.....	2
2.0 Introduction	5
2.1. Pollutant Deposition in the AOSR.....	5
2.2. Key Question of Oil Sands Deposition Monitoring	7
2.3. Objectives of the Evaluation and Integration Project.....	9
2.4. Existing Integration	10
2.5. Obstacles to Integration.....	11
2.6. References	12
3.0 Acidifying and Eutrophying Pollutants (N, S, Base Cations, P, Organic Acids).....	14
3.1. High-level Overview of Current Knowledge.....	14
3.1.1. Sources	14
3.1.2. Deposition.....	15
3.1.3. Receptor Concentrations.....	16
3.1.4. Ecological Effects.....	17
3.2. Significant Knowledge Gaps.....	18
3.3. Summary Figures	20
3.4. References	23
4.0 Mercury and Trace Elements	26
4.1. High-level Overview of Current Knowledge.....	26
4.1.1. Sources	26
4.1.2. Deposition.....	26
4.1.3. Receptor Concentrations.....	27
4.1.4. Ecological Effects.....	29
4.2. Significant Knowledge Gaps.....	31
4.3. Summary Figures	33
4.4. References	35

5.0 PACs	38
5.1. High-level Overview of Current Knowledge	38
5.1.1. Sources	38
5.1.2. Deposition	39
5.1.3. Receptor Concentrations	40
5.1.4. Ecological Effects.....	41
5.2. Significant Knowledge Gaps.....	42
5.3. Summary Figures	44
5.4. References	45
6.0 Recommendations for Integration and Additional Monitoring	48
6.1. Acidifying and Eutrophying Pollutants	48
6.2. Mercury and Trace Elements	50
6.3. PACs	52
6.4. Towards a Unified Deposition Program	52
Appendices	54
Appendix A – Maps of Monitoring Sites	54
Appendix B – Project Summary Tables	57
Appendix C – Acronyms and Abbreviations	76
Appendix D – Workshop Agenda and Participant List.....	79

Foreward

Since February 2012, the governments of Alberta and Canada have worked in partnership to implement an environmental monitoring program for the oil sands region. In December 2017 both governments renewed their commitment to working together with Indigenous communities in the region by the signing the *Alberta-Canada Memorandum of Understanding (MOU) Respecting Environmental Monitoring in the Oil Sands Region*. The MOU establishes the foundation for an adaptive and inclusive approach to program implementation ensuring that the program is responsive to emerging priorities, information, knowledge, and input from key stakeholders and Indigenous peoples in the region.

The Oil Sands Monitoring Program is designed to enhance the understanding of the state of the environment and cumulate environmental effects as a result of oil sands development in the region through monitoring and publically reporting on the status and trends of air, water, land and biodiversity. Its vision is to integrate Indigenous knowledge and wisdom with western science to design, interpret, assess, report and govern the program.

Canada and Alberta have provided leadership to strengthen program delivery, and ensure that necessary monitoring and scientific activities meet program commitments and objectives. The oil sands industry provides funding support for the program under the Oil Sands Environmental Regulation (Alberta Regulation 226/2013). Key findings and results from the program inform regional resource management decisions and importantly, are considered as an objective source of scientific interpretation of credible environmental data.

A mandated cornerstone of the program is the public reporting of data, status and trends of environmental impacts caused by development of oil sands resources. The Oil Sands Monitoring Program *Technical Report Series* provides an objective, and timely, evaluation and interpretation of monitoring data and information collected across environmental media of the program. This includes reporting and evaluation of emission/release sources, fate, effects and transport of contaminants, landscape disturbance and responses across theme areas including atmospheric, aquatic, biotic, wetlands, and community based monitoring.

1.0 Executive Summary

In November 2016, the Oil Sands Monitoring (OSM) co-chairs identified the need to evaluate and integrate atmospheric deposition monitoring conducted by OSM. There was a perceived lack of clarity on the existing monitoring and to what extent it was providing sufficient information to assess the impacts of deposition of OS emissions to surrounding ecosystems. A total of fourteen 2017/18 OSM-funded deposition projects were flagged for evaluation and integration. As a result, a 1-year joint AEP-ECCC focused study was created for 2017/18 with the following objectives: (i) summarize existing atmospheric deposition monitoring, (ii) identify significant knowledge gaps that should be, but are not, covered by an atmospheric deposition monitoring program, and (iii) develop recommendations to begin filling these knowledge gaps by integrating existing projects and/or implementing additional monitoring.

These objectives were addressed by first holding a series of knowledge-sharing webinars during May-July 2017 to disseminate information and raise awareness amongst project principal investigators (PIs). Second, a 1.5 day workshop was held from September 27-28 to discuss a draft of this document with the specific goals of: (i) finalizing significant knowledge gaps, and (ii) developing recommendations to begin addressing these gaps. A second draft of this report was circulated to workshop participants in November 2017 to finalize the report and help inform OSM project planning for 2018/19.

Technical details of 2017/18 OSM deposition projects are summarized in Appendix B, with a brief summary of the primary objective of each project given in Section 2.2. There is little, if any, duplication amongst existing monitoring projects. It must be recognized that some projects (e.g., A-LTM-3-1718 and B-MD-12-1718; subcomponents of A-MD-2-1718) are already intimately integrated (e.g., co-located sites, defined data uses and data sharing, complementary measurements), with details given in Section 2.4. There are also synthesis reviews that are being, or will be, prepared to summarize work-to-date on broad topics in OSM (e.g., PACs, mercury, forest health). These parallel reviews contribute to integration and help to develop informed strategies for future work. Section 2.5 discusses programmatic and logistical obstacles that have hindered previous OSM deposition integration efforts, including: imposed travel restrictions, limited work plan flexibility, proposal-based project planning, and lack of formal mechanisms to integrate projects at early stages.

To identify significant knowledge gaps, the “current level of knowledge” was assessed for: (i) sources, (ii) deposition patterns, and (iii) receptor loadings for each pollutant class (acidifying/eutrophying, Hg/trace elements, and PACs). The assessment was conducted using the peer-reviewed literature, relevant technical reports, and project plans. Summary Table 1 highlights the current level of knowledge (from “very poorly understood” to “very well understood”) for each aspect of each pollutant class. It should be noted that the number scale associated with each pollutant/aspect is a somewhat subjective and qualitative assessment.

Disclaimer: The content of this report reflects information as of November 2017. There have been modifications to several OSM projects detailed in this report with approval of the 2018-2019 Ambient Monitoring Environment Monitoring Plan for Oil Sands Development. As well, subsequent data analyses have also provided additional insights on atmospheric deposition monitoring. Information in this report should be considered within this context. A follow-up Deposition Integration Workshop will be held in fall 2018 to leverage this report to further inform the integration of deposition monitoring in the 2019-2020 Ambient Environment Monitoring Plan.

Summary Table 1: Current Level of Knowledge

Pollutant Class	Sources	Deposition Patterns	Receptor Loadings	Effects Observed or Suspected?
Acidifying and Eutrophying	SO _x (5) NO _x (4) NH ₃ (2) Base Cations (2) Organic Acids (1) P and “other” N (1)	Total S (4) Total N (2) Total Base Cations (2) Total Org Acids (1) Total P (1)	Terrestrial (4) Aquatic (3) Wetlands (2)	Acidification – no, except in rivers during spring freshet and some regional model simulations Alkalization – evidence in shallow lakes and terrestrial vegetation; mitigates acidification; likely from base cations Eutrophication – some evidence for ecosystem changes in forests and wetlands due to N or base cations
Hg and Trace Elements	Speciated Hg (2)	Elements (2) Total Hg (2) Methyl Hg (1)	Aquatic Hg/TM (4) Terrestrial TM (3) Wetland TM (2) Terrestrial Hg (2) Wetland Hg (2)	Hg – not usually, most observations are below available guidelines; there are limited data for wildlife Trace Elements – possibly, some observations of elements are sometimes above guidelines for soil, snowmelt, and water; there are limited data for wildlife
PACs	Speciated PACs (3)	Speciated PACs (4)	Aquatic (3) Terrestrial (3) Wetlands (1)	PACs – yes, enhanced levels observed in wolves, moose, caribou, birds; negative effects observed in otters but not Daphnia; some parent PAHs exceed soil and sediment guidelines; no guidelines for alk-PAHs or DBTs

- 1 = Very Poorly Understood (no monitoring, or limited number of contradicting studies)
 2 = Poorly Understood (little/incomplete monitoring, but generally consistent findings)
 3 = Somewhat Understood (some monitoring, and generally consistent findings)
 4 = Well Understood (significant monitoring, and generally consistent findings)
 5 = Very Well Understood (substantial monitoring, with clear and consistent findings)

Summary Table 1 forms the foundation for the identification of knowledge gaps. It provides a clear picture of areas where there is an incomplete understanding of a given aspect (i.e., sources, deposition, or receptor loadings) for a given pollutant class. However, for a lack of knowledge to be significant, there must also be evidence for a suspected or observed effect that arises from deposition of that pollutant. With this in mind, the following were identified as significant knowledge gaps that are not currently being adequately addressed by OSM deposition monitoring:

1. Magnitude and spatial patterns of base cation deposition
2. Sources of ammonia (NH₃) and poor understanding of bi-directional exchange
3. Magnitude and spatial patterns of total N deposition

4. Magnitude of regional (>150 km) deposition of N, S and base cations
5. Disseminating deposition surfaces (maps) to PIs investigating ecosystem effects
6. How to interpret ion exchange resin (IER) measurements
7. Sources of trace elements
8. Magnitude of fugitive dust emissions
9. Spatial and temporal patterns of fugitive dust deposition
10. Spatial and temporal patterns of wet deposition for mercury (Hg) and trace elements
11. Degree of exposure of sentinel biota to Hg and trace elements
12. Lack of ecological risk assessments for Hg, trace metals, and PACs
13. Lack of use of Standard Reference Materials (SRMs) for PACs measurements
14. Magnitude and speciation of PACs sources
15. Lack of understanding on relationship between PACs deposition and receptor concentrations

The details and rationale for these significant knowledge gaps are provided in Sections 3.2, 4.2 and 5.2. Some of these gaps (e.g., gaps #1, #2, #3, #5, and #7) can be at least partially addressed by integrating existing projects through activities such as data sharing and analyses. However, some of these knowledge gaps (e.g., gaps #8, #9, #10, and #13) will require adjustments to existing projects and/or additional monitoring. Recommendations for beginning to address these gaps are given in Sections 6.1 to 6.3. The recommendations listed above are intended to address gaps, and do not specifically address potential redundancies or extraneous monitoring.

This evaluation and integration project outlined in this report is a first step towards creating a unified OSM atmospheric deposition monitoring program. The report has summarized existing monitoring, significant knowledge gaps, and recommendations for beginning to fill those gaps. Section 6.4 provides a suggested path forward for transitioning from 14 somewhat disjointed deposition focused studies towards a more unified and coherent core long-term deposition monitoring program. A key element in developing a more unified and integrated deposition program under OSM is the on-going promotion of workshops and special conference sessions dealing with the oil sands. This will allow researchers to better recognize complementary areas for data analysis and future project opportunities across all disciplines.

2.0 Introduction

2.1 Atmospheric Pollutant Deposition in the AOSR

Atmospheric pollutants can be deposited via precipitation (wet deposition) or by turbulent exchange/settling (dry deposition). Wet deposition is calculated as the product of precipitation volume and pollutant concentration. Precipitation samples are typically collected on daily-to-weekly timescales using a pre-cleaned container and conditional sampling (i.e. the container is sealed during dry periods to prevent dry deposition), followed by quantification of pollutant concentrations in a lab (Vet et al., 2014). On the other hand, measuring dry deposition requires sophisticated methodology unsuitable for monitoring networks, so it is often inferred (modelled) as the product of the atmospheric concentration and deposition velocity (Wesely and Hicks, 2000). The deposition velocity describes the rate at which a pollutant dry deposits from the atmosphere and is empirically calculated based on turbulence parameterizations, pollutant/surface properties, as well as solubility and chemical reactivity (for gases), or size, density and shape (for particles) (Wesely, 1989; Zhang et al., 2001, 2003). Bulk deposition is the sum of wet deposition and some difficult-to-predict fraction of dry deposition. Bulk samples are usually collected using a container that is continuously open to the atmosphere. Total deposition is simply the sum of wet and dry deposition. Throughout this document, “deposition” refers strictly to “atmospheric deposition”.

Accurately quantifying total deposition in remote environments is challenging due to the need for (i) frequent site access to collect precipitation samples, (ii) power to run continuous air pollutant analyzers and meteorological equipment, and (iii) shelter to house sensitive instrumentation. However, these challenges can be at least partially mitigated by using passive air/bulk samplers and/or modelling. Uncertainty in measurements of wet deposition are believed to be small (from $\pm 10\%$ up to a factor of 2) compared to dry deposition (typically greater than a factor of 2) (Amodio et al., 2014; Mohan et al., 2016; Wu et al., 2018).

Pollutant deposition can have harmful effects on ecosystems and biota, hence quantifying deposition and understanding its effects is an important component of OSM. These impacts vary based on the pollutant, receptor and deposition pathway. Pollutants that can elicit an ecosystem or biotic response through deposition can be broadly classified into the following categories: acidifying pollutants, eutrophying pollutants, trace elements, and polycyclic aromatic compounds (PACs) (Wright et al., 2018).

Acidifying pollutants can harm sensitive biota by decreasing media pH (e.g., soil, lakes) and enhancing solubility of potentially toxic elements (Jacob, 1999). The major acidifying pollutants are oxidized S (e.g., SO_2 , aqueous SO_4^{2-} , particulate SO_4^{2-}), oxidized N (e.g., HNO_3 , aqueous NO_3^- , particulate NO_3^-), and reduced N (e.g., NH_3 , particulate NH_4^+). However, other compounds such as organic acids or hydrochloric acid can also contribute to acid deposition. Pollutants such as base cations (e.g., Ca_2^+ , Mg_2^+ , K^+) can neutralize acidifying pollutants and mitigate or prevent receptor acidification. Hence, assessing the effects of acidifying pollutants requires quantifying a wide range of pollutants in multiple phases. Facilities in the AOSR are known to be large sources

of acidic precursors (SO_2 and NO_x) and base cations (NPRI, 2015). In addition, large portions of soil in the AOSR have low weathering rates making them especially susceptible to acidification (Whitfield et al., 2010). Hence, there is a need to monitor the deposition and potential effects of acidifying and alkalizing pollutants in the region.

Eutrophication is the addition of excessive nutrients to an ecosystem and can result in undesirable changes in ecosystem function and structure (Smith et al., 1999). Typically, nitrogen (N) and phosphorus (P) are the limiting nutrients for plant biomass in terrestrial and aquatic ecosystems (Schlesinger, 1991; Smith, 1998). Quantifying total N deposition is challenging due to the many different species and biogeochemical transformations that occur in the environment. Since AOSR facilities are significant sources of NO_x and possibly NH_3 or other nutrients, there is a need to investigate eutrophying (nutrient) deposition and potential effects (e.g., Clair and Percy, 2015).

Trace elements are loosely defined as elements present in small but measureable quantities. Some of these elements can lead to toxic effects for biota. Thirteen trace elements are listed as priority pollutant elements by the US EPA (Sb, As, Be, Cd, Cr, Cu, Pb, Hg, Ni, Se, Ag, Tl, and Zn) due to their ubiquity and toxic effects at moderate-to-low concentrations. In particular, methyl mercury (MeHg) is an especially toxic form of Hg and can biomagnify in food webs (Lehnherr, 2014). With the exception of Hg, trace elements are almost exclusively emitted to the atmosphere in the particle-phase. On the other hand, Hg exists in both the gas (either elemental Hg^0 or oxidized Hg^{II}) and particle-phase (Hg^{II}), and can undergo biogeochemical transformations (Lindberg et al., 2007; Zhang et al., 2017). Industrial AOSR activities are known to release significant quantities of trace elements to the atmosphere (NPRI, 2015), hence understanding the deposition and possible effects to surrounding ecosystems is important.

PACs are a class of compounds consisting of fused aromatic rings that exist in both the gas and particle-phase. Subclasses of PACs include parent (unsubstituted) PAHs, alkylated PAHs (alk-PAHs), and dibenzothiophenes (DBTs). Certain PACs are toxic and carcinogenic at relatively low concentrations which have spurred significant environmental monitoring and toxicological research in recent years (WHO, 2010). Since the AOSR facilities are a known source of PACs, there is motivation to study the deposition and effects of these potentially toxic compounds.

Substantial effort has already been invested to understand whether pollutant deposition is negatively impacting ecosystems in the AOSR, and details of these efforts are discussed in subsequent sections. For convenience, this project broadly categorizes pollutants into two classes: (i) acidifying/eutrophying pollutants, and (ii) trace elements/PACs, because these classes typically affect ecosystems through different mechanisms (i.e., broad ecosystem-wide changes versus direct toxic effects in biota). Furthermore, monitoring of specific pollutants within (i) or (ii) are usually related (e.g., similar methodology, co-located sites, monitored by the same project). However, it must be recognized that any observed ecosystem effect cannot be assessed in such a fragmented fashion.

2.2 Key Question of Oil Sands Atmospheric Deposition Monitoring

Since the inception of Joint Oil Sands Monitoring (JOSM) in 2012, there has not been a single unified framework or program for monitoring pollutant deposition in the AOSR. Instead, deposition monitoring has been conducted through numerous individual focused studies (FS) and long-term monitoring (LTM) programs that typically target one or two pollutant classes (e.g. acidifying, eutrophying, elements, PACs) and/or ecosystem type (e.g., lakes, wetlands, forests). One additional complication is that individual FS and LTM are managed and implemented by different organizations (e.g., WBEA, AEP, and ECCC).

Despite the lack of a unified OS deposition monitoring program, all these individual FS and LTM are essentially trying to address at least one aspect of the same overarching question:

“Could anthropogenic emissions and/or deposition be affecting ecological systems in the oil sands region?”

In order to answer this key question on a pollutant-by-pollutant basis, a source-to-sink monitoring approach is required to gain sufficient knowledge about: (i) contaminant sources, (ii) deposition patterns, (iii) exposure pathways and concentrations in receptors including fluxes, and (iv) ecological effects (or lack thereof). Ultimately, the purpose of an OS deposition monitoring program should be to inform management actions that protect ecosystems from negative impacts. Emphasis should be placed on geographic areas and/or ecosystems with suspected or observed effects.

Fourteen OSM deposition-related projects have been flagged for consideration in this evaluation and integration project. Tables 2.1 (deposition-centric) and 2.2 (effects-centric) lists the primary deposition-related objective for each project. The majority of these projects have additional objectives that are not listed here (e.g., evaluating emission inventories, assessing air quality).

Table 2.1. Objectives for 2017-18 OSM Deposition-centric Projects

Project Title	PI	Primary Objective
Deposition and Effects – Passive PACs (A-MD-2)*	Tom Harner (ECCC)	Determine spatiotemporal trends for PACs in air and “baseline” air concentrations; inform deposition modelling and assess toxicity indicators for air
Deposition and Effects – Hg (A-MD-2)*	Alexandra Steffen (ECCC)	Understand the factors influencing Hg air concentrations including: (i) sources, (ii) transport, and (iii) transformation
Deposition and Effects – Enhanced Sites (A-MD-2)*	Ewa Dabek (ECCC)	Collect long-term air measurements of speciated PACs, elements, speciated PM _{2.5} , reactive gases and VOCs to help assess sources, transport, and transformations of these pollutants
Deposition and Effects – N and S (A-MD-2)*	Jason O’Brien (ECCC)	Measure the impact of Oil Sands N and S emissions and transformations on air concentrations and deposition downwind (~350 km away)
Deposition and Effects – Modelling (A-MD-2)*	Leiming Zhang (ECCC)	Produce gridded air concentration and total deposition maps for 43 PACs and trace elements
Deposition Monitoring in Forests (A-LTM-3)	Sanjay Prasad (WBEA)	Monitor air concentrations and deposition of acidifying and eutrophying pollutants at remote forest health sites; fully integrated with B-MD-12
Deposition to Lakes and Snowpacks (A-MD-9)	Jane Kirk (ECCC)	Identify OS sources for PACs/elements, and quantify deposition patterns/trends, as well as snow melt fluxes; integrated with some A-MD-2 components
OS Air Emissions, Transformation and Fate (A-MD-4)	Shao-Meng Li (ECCC)	Quantify OS emissions of SO _x , NO _x , VOCs, dust, GHGs, NH ₃ , and Hg, as well as assess transformations and fate by evaluating and improving models

*A-MD-2 components were amalgamated into one FS project plan in 2015 to better reflect and facilitate the integration between components

Table 2.2. Objectives for 2017-18 OSM Receptor-centric Projects

Deposition Project	PI	Primary Objective
Forest Health Monitoring (B-MD-12)	Sanjay Prasad (WBEA)	Determine if there are measurable effects on the most sensitive forest receptors (jack pine on sandy soil) by assessing correlations between forest parameters and pollutant deposition; fully integrated with A-LTM-3
Acid Sensitive Lakes Monitoring (A-MD-8)	Colin Cooke (AEP)	Monitor water quality parameters (e.g., pH, ions, elements, DOC, chl _a) annually in 50 shallow lakes and assess spatiotemporal trends
Wetland Ecosystem Monitoring (WL-MD-10)	Danielle Cobbaert (AEP)	Establish monitoring needs, objectives, and sites for an OSM wetland LTM program and begin measurements to correlate wetland parameters with measured/estimated pollutant deposition
Amphibian and Wetland Health (WL-MD-11)	Bruce Pauli (ECCC)	Assess wetland/ecosystem health and potential impacts from Oil Sands by monitoring PACs and elements in sentinel species (wood frogs) and water/sediments; integrated with A-MD-2 PACs passives
Colonial Waterbirds Monitoring (B-MD-8)	Craig Hebert (ECCC)	Determine if the Oil Sands are a source of Hg and PACs to waterbird eggs and assess spatiotemporal patterns; integrated with A-MD-2 PACs passives
Wildlife Contaminants and Toxicology (B-MD-9)	Philippe Thomas (ECCC)	Determine if negative health effects in otters are occurring and, if so, are linked to OS activities by analyzing alk-PAHs and stress hormones in otter feces

2.3 Objectives of the Evaluation and Integration Project

The direction for this 2017/18 project (A-MD-6 Evaluation and Integration of Deposition Studies) came from the OSM Co-Chairs in November 2016, with the project to be completed in sufficient time to inform 2018/19 project planning. The project was prompted by a lack of clarity on the full scope of OSM deposition work and the degree to which projects are and can be integrated. As a result, the primary objectives for this project are to:

1. Summarize current (2017/18) deposition and effects monitoring funded by OSM, as well as existing integration and obstacles to integration
2. Identify any significant scientific knowledge gaps that hinder our understanding of the impact of OS emissions on surrounding ecosystems

3. Provide recommendations for improved integration and/or future monitoring to address knowledge gaps and maximize benefits from resource sharing

The essence of this project is to collectively evaluate existing FS and LTM deposition projects to determine the extent to which the key OSM question is being addressed (see section 2.2), followed by recommendations for integration and/or additional monitoring to better address the key question.

Three knowledge-sharing webinars were held in May-July 2017 for PIs to present pertinent details of their FS and LTM studies. These webinars laid the foundation for summary tables (Appendix B) which list the pertinent details of each project (**Objective #1**). Information from the webinars and focused discussions during a two-day workshop in September 2017 were used to collaboratively generate a high-level overview on the current state of knowledge of (i) sources, (ii) deposition patterns, (iii) receptor concentrations, and (iv) ecological effects for each pollutant class (Sections 3.1-5.1) in order to identify knowledge gaps (Sections 3.2-5.2, **Objective #2**). Specific actions for integrating existing projects and areas requiring additional monitoring (Section 6) were collaboratively developed at the workshop in order to address significant knowledge gaps and maximize benefits from resource sharing (**Objective #3**).

In the context of this project, a “significant knowledge gap” refers to a topic where our understanding is both incomplete and hindering our ability to address the key monitoring question (see section 2.2). Therefore, for a knowledge gap to be “significant”, there must either be observed (through monitoring) or suspected (through evaluating receptor concentrations against a literature threshold) ecological effects related to that knowledge gap. For example, if the deposition of pollutant XYZ is greatly enhanced near OS facilities but there are no observed or suspected effects, then a poor understanding of the sources of XYZ is not a “significant knowledge gap” for deposition monitoring. Essentially, the focus of deposition monitoring and modelling should be to provide the necessary data to allow for the assessment of observed or suspected ecological effects.

2.4 Existing Integration

Several projects listed in Tables 2.1 and 2.2 are already partially or fully integrated. This section provides detail on the extent of existing integration, and is supplemented by summary figures in Sections 3.3 to 5.3 that provide a visual layout of how existing projects are integrated as well as what areas (i.e., sources, deposition, receptors, and/or effects) each project monitors. In addition, previous efforts to integrate OSM deposition projects are briefly discussed in the section.

Although the project B-MD-12 (Forest Health Monitoring) and A-LTM-3 (Deposition Monitoring in Forests) are separate project plans, the projects are fully integrated. B-MD-12 was designed to monitor terrestrial ecological indicators most sensitive to acidifying pollutant deposition, while A-LTM-3 was designed specifically to provide the necessary deposition data to assess the impact of acidifying deposition on the B-MD-12 indicators. Deposition monitoring sites are co-located with forest health and edge sites so that various ecosystem parameters can be assessed against measured deposition of acidifying and eutrophying pollutants.

Various components of A-MD-2 (Deposition and Effects) are also integrated with themselves, and other projects: WL-MD-11 (Amphibian and Wetland Health), B-MD-8 (Colonial Waterbirds Monitoring), and A-MD-4 (Air Emissions, Transformation, and Fate). Specifically, linkages include co-location of air and deposition monitoring of Hg, trace metals and PACs, as well as sharing of air monitoring data for PAC deposition model development within A-MD-2 (Deposition and Effects). Deposition and/or air concentration data are also shared with WL-MD-11 and B-MD-8 for investigating linkages between deposition and concentrations in biota.

Data from A-MD-9 (Deposition to Lakes and Snowpacks) and A-MD-4 (Aircraft Component) have been used to evaluate the GEM-MACH model for deposition of Hg, N, S, and base cations. In turn, the GEM-MACH model has been used to support A-MD-9 (Acid Sensitive Lakes Monitoring) to assess the impact of oil sands Hg emissions on Hg concentrations monitored in acid sensitive lakes.

There have also been three separate 1-day OS deposition workshops hosted at Environment and Climate Change Canada in June 2012, June 2013, and March 2016. Each previous workshop involved presentations from PIs on project goals, methods, results, outcomes, and future plans. The intent was to share knowledge to facilitate integration and discuss scientific direction. The major outcome for these previous deposition integration workshops were brief reports (~1-5 pages) that summarized the presentations and key discussion points. The March 2016 workshop also included the following recommendations that were “identified by the group as very important for future work and integration”:

1. Increase opportunities to communicate results amongst PIs through workshops and/or conference special sessions
2. More co-located measurements across different media (e.g., snow, air, water, lichens)
3. Enhance integration between deposition and ecosystem effects studies by allowing ecosystem studies to be the driver of deposition measurement site locations
4. Perform comparison between different methods/labs that measure the same contaminants
5. Measure particles larger than 10 µm in diameter (i.e. monitor TSP)

2.5 Obstacles to Integration

There are several obstacles within JOSM that hinder deposition integration, but which are not unique to deposition projects. It is important to note that these overarching obstacles have impacted previous deposition integration efforts, while recognizing they persist and hamper this current effort.

- **Imposed travel restrictions:** creates an inability to hold face-to-face science workshops, which can be an effective mechanism to promote integration by increasing scientific awareness amongst PIs, discussing work plans, and setting program priorities. Scientific awareness is particularly crucial for a topic as technically diverse as deposition monitoring. In contrast to more focused JOSM themes (e.g., lotic monitoring), PIs for deposition-related

projects typically do not interact as much due the broad variety of scientific backgrounds (e.g., atmospheric, aquatic, biodiversity, wetland, chemical fate modelling).

- **Limited work plan flexibility:** during the JOSM implementation phase (2012-2015/16), work plans were more-or-less fixed. Requests to add additional activities or linkages were often met with postponement until implementation was complete. Reduced flexibility can make it difficult to adapt work plans to bolster integration between projects.
- **Proposal-based project planning:** the current planning system relies on PIs submitting individual work plans (proposals) for approval, often with limited direction on the monitoring priorities of OSM or incentive to integrate. Such a proposal-based system does not intrinsically encourage integration, and risks creating a competitive funding environment.
- **Lack of formal mechanisms for integrating projects at early stages:** facilitating the sharing of draft project plans and preliminary data would help increase scientific awareness amongst PIs and make it easier to integrate projects prior to, or shortly after, implementation. Currently, as witnessed at the September 2017 workshop, PIs are sometimes unaware of relevant data or project plans until after they are publically released.

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3.0 Acidifying and Eutrophying Pollutants

3.1 High-Level Overview of Current Knowledge

3.1.1. Sources

The location and magnitude of SO₂ sources in the region are very well understood. Nearly all (>99%) of the SO₂ emissions from Oil Sands operations come from point sources and ~80% are directly quantified using Continuous Emissions Monitoring Systems (CEMS) (JOSM, 2016; Percy, 2013). Natural sources of SO₂ and direct emissions of pSO₄²⁻ are negligible in the region.

The location and magnitude of NO_x sources in the region are well understood. The majority of anthropogenic NO_x in the region comes from stacks and mine fleet vehicles, and is usually quantified with CEMS or emission factors (JOSM, 2016). Non-industrial anthropogenic NO_x emissions are ~5% of industrial emissions and include commercial/residential heating and traffic (JOSM, 2016; Davies et al., 2012; Percy, 2013). Lightning and natural soils can be significant NO_x sources but are poorly constrained for the region. The oxidation of the emitted NO_x and direct emissions from wildfires are likely the major sources of HNO₃ and pNO₃⁻.

The location and magnitude of NH₃ sources in the region are poorly understood. Although some NH₃ is emitted by point sources (JOSM, 2016), the majority is emitted from difficult-to-quantify area sources such as wildfires and plants/soil (Whaley et al., 2018). Whaley et al., (2018) estimated that bi-directional exchange (re-emission from plants/soil) contributed ~50% of NH₃ across the region, although this is the only study to date to consider bi-directional exchange in the area. NH₃ emissions from mine fleets, tailings ponds, and lakes have not been quantified (JOSM, 2016).

The location and magnitude of primary emission sources of “other N species” (e.g., N₂O₅, PAN, PPN, HONO, amines, organic N) are very poorly understood, with the exception of HNCO (Liggio et al., 2017a). Previous studies have implicitly assumed them to be insignificant compared to sources of NO_x, NH₃ and HNO₃/pNO₃⁻ (e.g., Clair and Percy, 2015; Hsu et al., 2016; Fenn et al., 2015; Wieder et al., 2016).

The magnitude of base cation sources is poorly understood. The majority of base cations in the region are emitted as wind-blown dust from area-wide fugitive sources such as mine faces and haul roads (Watmough et al., 2014). Although the locations of these sources are evident, there are large uncertainties for the (i) magnitude, (ii) chemical composition, and (iii) particle size distribution of Oil Sands dust emissions (JOSM, 2016).

The location and magnitude of organic acid sources are poorly understood. According to the 2013 intensive aircraft study (Liggio et al., 2017b), ~90% of gaseous low molecular weight organic acids (LMWOA, <C10) were formed from photochemical reactions of precursor VOCs. Large uncertainties exist for: (i) precursor VOC emissions, (ii) particulate organic acid burden, (iii) longer-chain (>C10) organic acids, and (iv) temporal variability of organic acid sources. LMWOA

emission/production related to Oil Sands activities is comparable to the magnitude of Oil Sands SO₂ emissions (~200-300 t day⁻¹).

The location and magnitude of total phosphorus sources in the region are very poorly understood. There is some evidence that the major source of total atmospheric P is fugitive dust from disturbed landscapes (Mullan-Boudreau et al., 2017; Summers et al., 2016). However, the speciation, magnitude, and temporal variability of P emissions in the region have yet to be investigated.

3.1.2. Deposition

Total S deposition in the AOSR (<100 km) is well understood and dominated by dry SO₂ and wet SO₄²⁻ (Clair and Percy, 2015). Regional maps of dry SO₂ deposition (Hsu et al., 2016) and bulk SO₄²⁻ deposition (Fenn et al., 2015; Wieder et al., 2016) have been made by interpolating observations, although wet SO₄²⁻ deposition data are limited to Fort McKay and Fort McMurray (Lynam et al., 2015). There is a good understanding of seasonal patterns of SO₂ dry and SO₄²⁻ bulk deposition (Clair and Percy, 2015).

Total N deposition in the region is poorly understood, in part because of the plethora of reactive N-species and confounding processes (e.g., NH₃ bi-directional exchange). Total N deposition has been reported to be dominated by reduced-N (dry NH₃ > wet NH₄⁺) but with significant contributions from oxidized-N (dry HNO₃, wet NO₃⁻, dry NO₂) (Fenn et al., 2015; Hsu et al., 2016). Previously estimated NH_x dry deposition in the AOSR ecosystems may need to be adjusted lower considering the bi-directional exchange feature of NH₃. As shown in Whaley et al., (2018), in some areas ecosystem NH₃ emissions were larger than NH₃ dry deposition, although the net NH_x dry deposition flux was still downward over most of the domain. Furthermore, the deposition of “other N species” (e.g., N₂O₅, HNCO, PAN, PPN, HONO, amines, organic N) has not been evaluated, but at other rural Canadian sites has constituted up to 35% of dry deposition (Zhang et al., 2009). Regional maps of interpolated dry NH₃/HNO₃/NO₂ (Hsu et al., 2016), bulk NH₄⁺/NO₃⁻ (Fenn et al., 2015; Wieder et al., 2016), and total snowpack N/DIN (Summers et al., 2016) deposition have been developed. There is still a limited understanding of key components of total N deposition, including: (i) wet deposition, (ii) bi-directional NH₃ exchange, and (iii) contribution of “other N species”.

Total base cation deposition in the region is poorly understood, primarily due to a lack of measurements needed to calculate dry deposition. Bulk deposition of Ca₂⁺, Mg₂⁺, and Na⁺ are measured using ion exchange resins (IERS; e.g., Fenn et al., 2015; Watmough et al., 2014) to produce interpolated maps of bulk base cation deposition (Clair and Percy, 2015). There is still a very limited understanding of (i) wet base cation deposition, and (ii) speciated, size-resolved PM measurements. Size distribution of base cations were measured for short periods at other rural Canadian sites (L. Zhang et al., 2008)

Total P deposition in the region is very poorly understood since there are no ongoing, year-round measurements of dry, wet, bulk or total P deposition. Springtime snowpack measurements showed that wintertime total P deposition was enhanced within ~50 km of OS facilities, but bioavailable forms were not (Summers et al., 2016). There is a very limited understanding with

regards to: (i) pathway (dry vs. wet), (ii) temporal variability, (iii) speciation, and (iv) magnitude of P deposition.

Total organic acid deposition is very poorly understood. The only major study to investigate atmospheric organic acids in the AOSR highlighted the large uncertainties on organic acid deposition and effects (Liggio et al., 2017b).

One of the most extensive deposition datasets in the region are the bulk IER measurements. IERs are advantageous because they do not require power and can be deployed for long periods. However, bulk open (no canopy) and throughfall (below canopy) methods can significantly and unpredictably differ from wet/total N, S, and base cation deposition (e.g., Blake and Downing, 2009; Fenn et al., 2013; Liu et al., 2006, 2015; Y. Zhang et al., 2008). Previous studies in the region have implicitly or explicitly assumed throughfall/open IERs to represent total/wet deposition (Clair and Percy, 2015; Fenn et al., 2015; Whitfield and Watmough, 2015). However, the only co-located study (Hsu et al., 2016) in the region to date has shown poor correlation of IER throughfall to measured total deposition of NO_3^- ($r=0.20$, $p=0.66$, $\text{IER/total} = 0.42$) and SO_4^{2-} ($r=0.75$, $p=0.04$, $\text{IER/total} = 6.21$), although reasonable correlation with NH_4^+ ($r=0.82$, $p=0.01$, $\text{bulk/total} = 0.92$). Since the comparison was done at AMS1, the relevance for remote forest sites might be limited. It remains unclear whether IER measurements can adequately represent total or wet deposition for inorganic S, N, or base cations in the region.

3.1.3. Receptor Concentrations

Receptor concentrations of major acidifying and eutrophying species in terrestrial ecosystems are well understood. Numerous parameters are measured in soil (pH, total N, total S, SO_4^{2-} , NO_3^- , NH_4^+ , available P, base cations, Al), jack pine needles (total N, total S, total P, SO_4^{2-} , base cations), and lichens (total N, total S, base cations) as part of the Forest Health Monitoring (FHM) program (Clair and Percy, 2015 and references therein). The FHM program has the ability to determine both spatial and long-term temporal patterns of relevant receptor concentrations in jack pine forests throughout the region, which have been determined to be the most sensitive terrestrial receptor for acidifying deposition.

Receptor concentrations of major acidifying and eutrophying species in aquatic ecosystems are somewhat understood, at least spatially. These species are routinely monitored in 50 shallow lakes (pH, SO_4^{2-} , NH_4^+ , NO_3^- , PO_4^{3-} , TN, TDN, TP, base cations) and numerous lotic systems throughout the region. Temporal coverage of river systems (several samples per year) are better than for lakes (one sample per year), although monitoring has been conducted for 20+ years in some rivers and lakes.

Receptor concentrations of major acidifying and eutrophying species in wetland ecosystems are poorly understood. Major species have been monitored previously in moss/lichens/shrubs/needles (total N, total S) and bog porewater (pH, SO_4^{2-} , NH_4^+ , NO_3^- , DIN, DON) in 4 ombrotrophic bog sites since 2009 and up to 19 sites intermittently by focused studies (Wieder et al., 2016a,b). The spatial extent of wetland monitoring is planned to increase through the ongoing development of a Wetland LTM program. Relative to terrestrial and aquatic monitoring, existing data sets are generally more limited both spatially and temporally.

3.1.4. Ecological Effects

There is little evidence of widespread acidification to date, likely due to a mitigating effect from concurrent base cation deposition. Several studies have observed deposition of base cations exceeding the sum of acidifying pollutants (typically considered to be SO_x , NH_x and NO_x/HNO_3) within tens of kilometres of OS facilities (Clair and Percy, 2015; Fenn et al., 2015; Watmough et al., 2014). This is consistent with a lack of evidence for regional-scale acidification of lakes (Andrews et al., personal communication; Curtis et al., 2010; Hazelwinkel et al., 2008; Laird et al., 2013) and soils (Clair and Percy, 2015; Jung et al., 2013). However, a recent study showed that five rivers in the region have experienced acidic episodes during 39% of snowmelt events over the last 25 years (Alexander et al., 2017). Wieder et al., (2016b) found higher H^+ and SO_4^{2-} in bog porewater closer to OS facilities. Clair and Percy (2015) also noted there is increasing soil S within ~20km of OS facilities. Makar et al., (2018) found that GEM-MACH simulations predict acid critical load exceedances occurring throughout the AOSR, with the spatial extent varying greatly depending on ecosystem type, critical load dataset, and correction factors (i.e. model-tuned by different observations). On the other hand, there is some evidence that excessive base cation deposition could be causing a shift in terrestrial vegetative communities (Clair and Percy, 2015) and an increase in lake pH (Andrews et al., personal communication), suggesting a possible alkalization effect throughout the region.

There is evidence that base cation and N deposition within ~50km of OS facilities are impacting terrestrial ecosystems. Specifically, Clair and Percy (2015) summarized the following relationships: (i) differences between soil microbial communities along the N+S bulk deposition gradient, (ii) elevated N, S, and Ca in jack pine needles within 20km of OS activities, (iii) positive (negative) correlation between elevated N/S/base cation bulk deposition and vascular plant (moss/lichen) cover and richness, and (iv) negative correlation between internode length and PAI. The authors suggest the most significant driver of terrestrial ecosystem shifts is likely base cation deposition followed by N eutrophication, and that there is a negligible impact from acidifying substances.

There is evidence that N deposition within ~50km of OS facilities is impacting wetland ecosystems. The following relationships have been observed with increasing proximity to OS facilities: (i) increased N/S in tissues of some lichens, mosses and vascular plants (Wieder et al., 2016b), (ii) enhanced vertical growth and NPP of moss (*Sphagnum fuscum*) (Wieder et al., 2016a), and (iii) increasing net C, N and S accumulation in peat over the last 25 years (Wieder et al., 2016a). Five consecutive seasons of artificial N-addition in a bog and poor fen have revealed the following as N-deposition increases: (i) down-regulation of biological N_2 -fixation (Vile et al., 2014), (ii) N-leaching at high N-loading ($25 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) (Vitt, 2016), and (iii) increased vascular plant biomass at high N-loading (Vitt, 2016). However, it is important to note that confounding effects from climatic factors (e.g., precipitation and temperature) and N_2 -fixation are likely impacting these observations (Vile et al., 2014; Wieder et al., 2016a).

There is no evidence that enhanced N or P deposition have caused eutrophication in aquatic ecosystems. Furthermore, a significant nutrient enrichment in aquatic systems due to OS development has not been detected. Although regional increases in aquatic primary productivity in shallow lakes have been observed, these changes are likely attributable to climate change as opposed to nutrient deposition (Summers et al., 2016; Mushet et al., 2017).

3.2 Significant Knowledge Gaps

Interpreting Ion Exchange Resin Data – IERs are a powerful monitoring technique due to their simplicity and ability to monitor bulk deposition in remote environments. However, there is a lack of clarity on how to interpret IER results – can these data be used to represent wet or total N, base cation and S deposition? Are spatial bulk deposition patterns representative of total deposition patterns? A co-location study at multiple CAPMoN (e.g., Pinehouse Lake and Flat Valley) and enhanced deposition sites could help clarify how IER data should be interpreted (i.e., IER bulk \approx total? IER bulk \approx wet? For what species? For which sites?).

Base Cation Deposition – there is evidence that base cation deposition is mitigating acidification (close to facilities), and possibly alkalizing shallow lakes as well as nutrifying jack pine ecosystems. However, annual/seasonal dry and wet base cation deposition is poorly constrained. A better understanding of dry and wet base cation deposition is required to determine whether fugitive dust from OS facilities is causing these effects. Furthermore, it would improve our ability to predict and model future emission scenarios and potential exceedances of acidic critical loads.

NH₃ Sources and Bi-directional Exchange – There is evidence that NH_x (=NH₃ + NH₄⁺) is the largest contributor to total N deposition within the AOSR. Furthermore, NH_x deposition is an important component of calculating acidic critical load exceedances. However, our understanding of NH₃ sources is incomplete (e.g., tailings ponds, mine fleet) and bi-directional exchange is poorly constrained. A better understanding of NH₃ sources and bi-directional exchange (i.e. which receptors have NH₃ emission > dry deposition) would significantly improve our currently incomplete estimates of total N deposition.

Total N Deposition – There is evidence that N deposition is altering wetland and terrestrial ecosystems. However, our knowledge of total N deposition is incomplete. It is unknown whether “other N species” (e.g., N₂O₅, HNCO, PAN, PPN, HONO, amines, organic N) are significant contributors to total N deposition and whether any of these species merit routine monitoring.

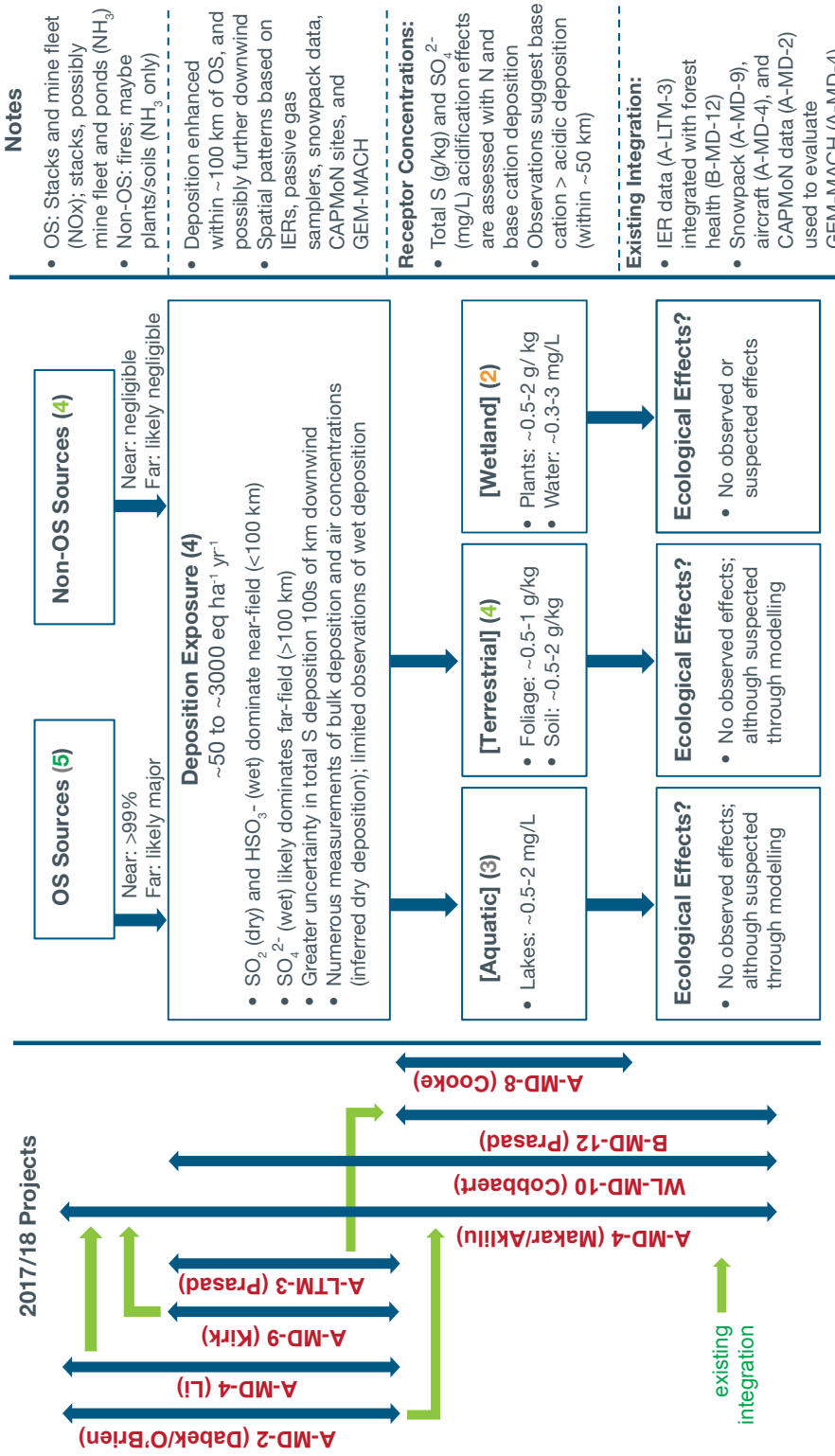
Regional (>150 km) Downwind N, S and Base Cation Monitoring – There is evidence from 2013 aircraft study that SO₂- rich plumes are advected 100s of km downwind into Saskatchewan. Since enhanced base cation (dust) deposition is likely limited to <50 km, there exists the potential for acidification further afield. However, there is limited monitoring of these species beyond the Alberta border at two CAPMoN sites (Flat Valley, SK and Pinehouse Lake, SK). It should be noted that base cation and precipitation measurements at these two CAPMoN sites are not supported by OS funding.

Disseminating Deposition Surfaces – There is a utility for deposition surfaces (maps) for N, S and base cations to support effects monitoring at sites that are not co-located with deposition monitoring. Although there is capacity for these deposition surfaces to be created (by e.g., Kriging interpolation, GEM-MACH modelling), there are currently no formal mechanisms to disseminate these data products to receptor-centric projects.

Wetland LTM – There is evidence that enhanced N deposition is altering wetland ecosystems, although there is a poor understanding of total N deposition and wetland concentrations/parameters. However, the development of a Wetland LTM is in progress (WL-MD-10) and its aim is to integrate with existing programs (e.g., B-MD-12). Hence, further discussion and recommendations pertaining to Wetland LTM are not detailed in Section 6.

3.3 Summary Figures

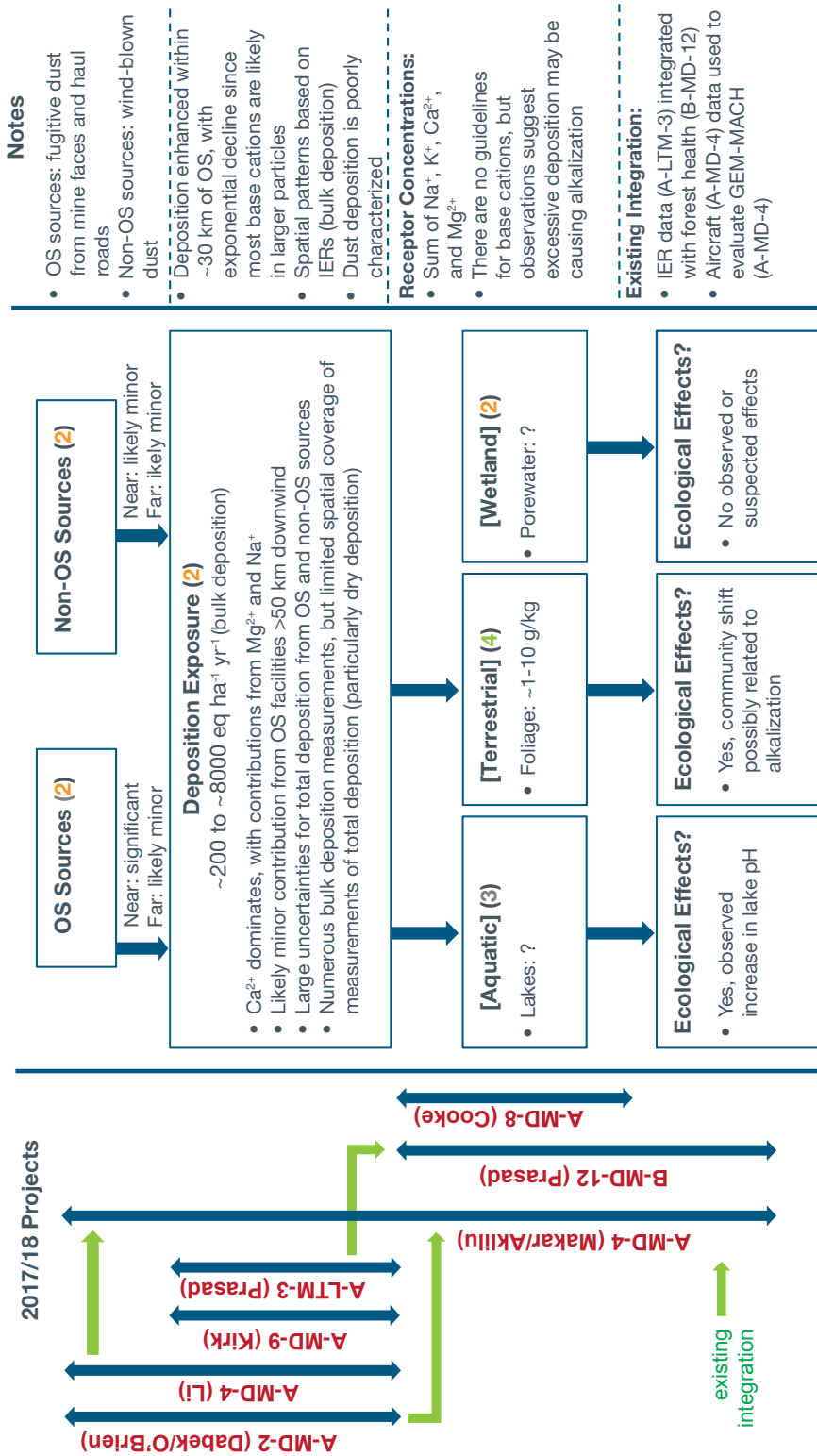
Could atmospheric S deposition from OS emissions be affecting ecosystems? Maybe



Deposition Monitoring Needs and Knowledge Gaps

- There is a moderate need for monitoring S deposition because of: (i) its an important component of acidic deposition, (ii) observations showing alkalization of some lakes and soils (see base cation summary figure), and iii) contrasting simulated acidification
- Significant knowledge gaps that hinder answering the key question:
 - Incomplete characterization of IER methodology – it is unclear what fraction of total S deposition is monitored by IERs
 - Large uncertainties on extent of base cation deposition – effects of S deposition depend on relative N+S and base cation deposition
 - Uncertainty on regional (>150 km) S deposition – mitigating base cation deposition is likely negligible at these distances

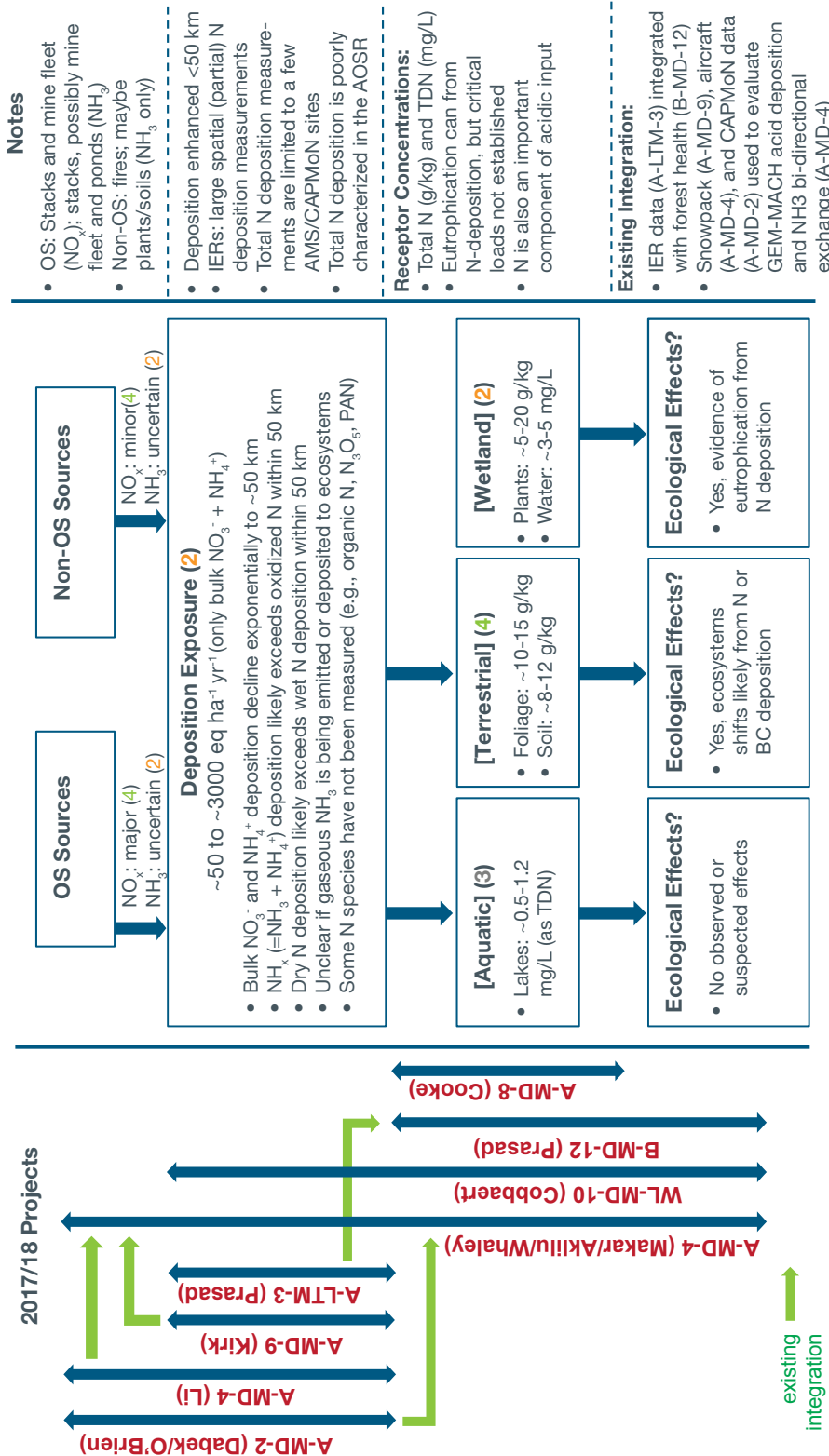
Could atmospheric base cation deposition from OS emissions be affecting ecosystems? Yes



Deposition Monitoring Needs and Knowledge Gaps

- There is a strong need for monitoring base cation (dust) deposition because of: (i) observed ecological effects likely linked to alkalization, (ii) significantly enhanced base cation (dust) deposition near OS facilities, and (iii) important role of base cations in evaluating acidic deposition
- Significant knowledge gaps that hinder answering the key question:
 - Incomplete characterization of IER methodology – it is unclear what fraction of total base cation deposition is monitored by IERs
 - Large uncertainties on magnitude and seasonality of total base cation deposition, particularly for dry deposition
 - Large uncertainties on magnitude of natural and anthropogenic emissions

Could atmospheric N deposition from OS emissions be affecting ecosystems? Yes



Deposition Monitoring Needs and Knowledge Gaps

- There is a strong need for monitoring N deposition because of: (i) observations showing ecological shifts possibly linked to N deposition, (ii) limited understanding of total N deposition, and (iii) N is an important component of acidic deposition
- Significant knowledge gaps that hinder answering the key question:
 - Incomplete characterization of IER methodology – it is unclear what fraction of total N deposition is monitored by IERs
 - Incomplete accounting of total N deposition (i.e., “other” N species not currently monitored)
 - Poor characterization of NH3 sources and bi-directional exchange (i.e., is NH3 dry deposition significant for all ecosystems?)
 - Uncertainty on regional (>150 km) N deposition – mitigating base cation deposition is likely negligible at these distances

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4.0 Mercury and Trace Elements

4.1 High-Level Overview of Current Knowledge

4.1.1. Sources

The location and magnitude of Hg sources are poorly understood. There is only one published study analyzing the sources of total gaseous mercury (TGM) at one site in the AOSR (Parsons et al., 2013). There are no published studies on the sources of oxidized Hg (i.e. gaseous oxidized Hg and particle-bound Hg), which are more susceptible to deposition and readily converted to methyl mercury (MeHg) that can be taken up by organisms. The one study suggests TGM concentrations are affected by variations in meteorological parameters, surface emissions over the region, and long range transport of emissions, instead of direct emissions from oil sands development. Forest fire emissions are responsible for elevated TGM episodes (Parsons et al., 2013). Hg releases from oil sands facilities are reported to the National Pollutant Release Inventory (NPRI; ECCC, 2017). All of the facility reported Hg emissions are attributed to point sources. The amount of Hg disposed in tailings is significantly larger than point source emissions; however, it is unknown whether tailing ponds are sources of Hg to air, soil or water bodies.

The location and magnitude of trace element sources are somewhat understood. Trace elements are predominantly associated with particulate matter (PM); therefore they are likely to be impacted by common sources. Primary sources of PM in the AOSR include fugitive dust (e.g., open-pit mining, heavy-duty vehicular traffic, petcoke dust, unpaved surfaces, forest soil), fossil fuel combustion (e.g., boilers, furnaces, vehicular emissions), forest fires, and wood combustion (Wang et al., 2015; Shotyk et al., 2016; Landis et al., 2017; Xing and Du, 2017). Based on receptor modeling results (Landis et al., 2017; Phillips-Smith et al., 2017), fugitive dust is a major source of crustal elements, e.g., Al, Ca, Fe, K, Si, and Ti. Oil sands upgrading processes contribute to S, Mo, V, Ni, As, Br and Pb. Biomass burning is a source of K, Zn, and Cd (Landis et al., 2017; Phillips-Smith et al., 2017). The facility emissions of 14 trace elements are reported to the NPRI (ECCC 2017). The majority of the reported releases are from point sources followed by direct discharge. Fugitive and non-point source releases make up a very small proportion in reports to the NPRI, which differs from receptor modeling results. V, Zn, Ni and Mn comprise most of the trace metal emissions from oil sands facilities. The amount of trace elements disposed in tailings is significantly larger than other types of releases; however, it is unknown whether tailing ponds are sources of trace elements to soil or water bodies.

4.1.2. Deposition

Total Hg deposition is poorly understood. There are a few short-term studies measuring Hg in bulk (wet + dry) deposition and in wet deposition in the AOSR during winter or spring season. Hg in bulk deposition and wet deposition are typically lower compared to other elements (Bari et al., 2014; Kirk et al., 2014; Lynam et al., 2015). Hg deposition decreases exponentially with distance from oil sands sources up to ~80 km (Kelly et al., 2010; Bari et al., 2014; Kirk et al., 2014). This spatial trend is also found for MeHg (Kirk et al., 2014). Hg and MeHg in snowpacks

are predominantly bound to particles, which likely explain the higher deposition at sampling sites near oil sands sources and the decreasing deposition with distance (Kirk et al., 2014).

Total trace elements deposition is poorly understood. Similar to Hg, only a few studies have measured trace elements in bulk deposition and wet deposition in the AOSR. Both crustal (e.g., Ca, Al, Fe, Mg) and anthropogenic (e.g., V, Ni, Zn, Ni) elements are found in deposition (Bari et al., 2014; Kirk et al., 2014; Lynam et al., 2015; Guéguen et al., 2016). The deposition of most of the trace elements decreases exponentially with distance from oil sands sources up to ~85 km (Kelly et al., 2010; Bari et al., 2014; Kirk et al., 2014; Guéguen et al., 2016; Mullan-Boudreau et al., 2017). This is observed for elements in particulate and dissolved-phases in snow (Kelly et al., 2010). However, there are some elements (e.g., Cd, Cl, Cr, Mn) with no spatial gradients in the deposition, which suggests the impact of local/regional sources rather than oil sands development (Kelly et al., 2010; Guéguen et al., 2016).

There is a lack of studies on dry deposition of Hg and trace elements and deposition modeling of these pollutants in the AOSR. Models can provide deposition estimates over a larger area and at remote locations with limited access. Dry deposition fluxes can be estimated using modeling approaches (e.g., inferential methods) given the technical challenges with dry deposition measurements (Zhang et al., 2016). These approaches can also quantitatively apportion deposition fluxes to sources and identify specific areas with elevated deposition fluxes which can inform ecosystem risk assessments.

4.1.3. Receptor Concentrations

Hg concentrations have been measured in ambient air, rivers, lakes, sediments and epiphytic lichens in the AOSR. In ambient air, the average TGM concentration in Fort McMurray is $1.45 \pm 0.18 \text{ ng m}^{-3}$ (Parsons et al., 2013), which is comparable to the average TGM at other Canadian sites and lower than the average TGM near the former copper smelter in Flin Flon, Manitoba (Cole et al., 2014). During forest fire episodes in the AOSR, average TGM increases to $1.73 \pm 0.34 \text{ ng m}^{-3}$ (Parsons et al., 2013).

Receptor concentrations of Hg in aquatic ecosystems are well understood. Hg concentrations in the Athabasca River near oil sands development and in tributaries affected by land disturbance are higher than upstream. Higher Hg concentrations are also found near the Athabasca Delta and Lake Athabasca (i.e. downstream of oil sands development) than upstream (Kelly et al., 2010). Sediment cores collected in the Peace Athabasca Delta (PAD) and Cold Lake, which are sites a few hundred kilometers away from oil sands surface mining areas, indicate Hg concentrations in sediment have been declining since the beginning of oil sands development (Wiklund et al., 2012; Skierszkan et al., 2013). Hg concentrations in sediment reached its maximum between 1965 and 1990 and declined afterwards, which differs from the increasing oil sands development since 1990 (Wiklund et al., 2012). Hg concentrations measured in surface sediments in AOSR lakes are low and similar to concentrations in lakes close to power plants in Alberta. Unlike deposition, no differences in the Hg concentrations are found for sediments near oil sands sources and farther away (Neville et al., 2014).

Receptor concentrations of Hg in wetland ecosystems are poorly understood. Sediment samples in natural and constructed wetlands indicate that Hg and MeHg in natural wetlands are lower than other boreal wetlands, but the percentage of MeHg is comparable. Despite the proximity of the constructed wetlands to bitumen upgraders, Hg and MeHg concentrations in the wetland sediments are low due to other factors, e.g. evasion of gaseous elemental Hg from the wetland and high mineral content of the constructed wetland resulting in reduced binding of inorganic Hg to sulfur-containing organic matter (Oswald and Carey, 2016).

Receptor concentrations of Hg in terrestrial ecosystems are poorly understood. Terrestrial Hg measurements are limited to epiphytic lichens in the AOSR and are similar to those measured at background locations in other parts of the world. In contrast, Hg concentrations in lichens are much higher within 1-8 km from chlor-alkali plants and found to correlate with TGM. However, TGM in the AOSR are not near the concentrations observed around chlor-alkali plants. Hg concentrations in lichens near oil sands sources are slightly lower than those sampled at farther distances (Blum et al., 2012).

Overall, most of the studies suggest that the oil sands development have not resulted in significant increases in Hg concentrations in ambient air, sediments, and lichens in the AOSR. However, the Athabasca River and tributaries have been affected by oil sands development. Establishing links between Hg sources and sinks is complicated because Hg can convert between elemental and oxidized forms in air, which deposit at varying distances from emission sources and at varying magnitudes. Models may be useful tools to apportion receptor concentrations to sources.

Trace element concentrations have been measured in ambient air, soil, rivers, lakes, sediments, lichens, moss, and peat cores in the AOSR. In ambient air, average concentrations of some trace elements near oil sands mining areas are lower than those measured at Canadian cities; however, Si, Ti, K, Fe, Ca, and Al are higher at the oil sands sites. In some instances, large peaks in S, Ba, Br, and Mn concentrations are observed, which suggests the impact of anthropogenic sources (Phillips-Smith et al., 2017).

Trace element concentrations in terrestrial ecosystems are somewhat understood. In soil, trace element concentrations are higher at sites near oil sands mining areas than sites outside this area. The concentrations for most of the trace elements are negatively correlated with distance from bitumen upgraders (Boutin and Carpenter, 2017). Trace element concentrations (As, Cd, Ni, and V) are slightly higher in soil near in-situ oil extraction in Cold Lake; however, the opposite trend is found for Pb (Skierszkan et al., 2013). Some of the element concentrations correlate with Fe near the in-situ oil fields suggesting the higher iron levels in the soil increase the adsorption of trace elements.

Al and V concentrations in lichens decline exponentially from oil sands sources; however, Mn is lower near oil sands sources and higher at distant sites. Pb concentrations in lichens follows an exponential decline within 50 km of oil sands sources, but large variability in concentrations are found at distant sites (Graney et al., 2012).

Trace element concentrations in aquatic ecosystems are well understood. Like Hg, most of the trace element concentrations in the Athabasca River, Athabasca Delta and Lake Athabasca downstream of oil sands development are higher than upstream sites. Higher concentrations are also found in tributaries with greater oil sands development (Kelly et al., 2010). Trace element concentrations, such as As, Cd, Sb and Pb, measured from lake sediment cores outside of oil sands surface mining areas (PAD and Cold Lake) have been declining or leveling off since the beginning of oil sands development (Wiklund et al., 2012; Skierszkan et al., 2013). Trace element concentrations in sediments peaked during the 1900-1950s and have since been declining similar to the trends in industrial emissions across North America (Wiklund et al., 2012; Skierszkan et al., 2013).

Trace element concentrations in wetland ecosystems are somewhat understood. Most of the trace element concentrations in moss samples collected in bogs are lower or equivalent to those at background sites. Only V, which is abundant in bitumen, is enriched in moss (Shotyk et al., 2014, 2016). For most trace elements including those abundant in bitumen (e.g. V, Ni, and Mo), the increases in concentrations in moss are largely due to increases in mineral dust levels caused by land disturbance, petcoke dust, and unpaved roads (Shotyk et al., 2014, 2016). Enrichment in trace elements (e.g., V, Ni, Mo, Ag, Cd, Sb, and Tl) in peat cores has been declining since the 1970s (Shotyk et al., 2017).

Overall, most of the studies suggest that the oil sands development has not significantly contributed to trace metal contamination in sediments, moss and peat in the AOSR. Among the trace elements, enrichment in bitumen-abundant elements like V and Ni requires further monitoring. Trace element concentrations in ambient air, soil and along the Athabasca River and its tributaries have been affected by oil sands development.

4.1.4. Ecological Effects

Hg in snowmelt and water near oil sands development or downstream of the development exceed Canada and Alberta water quality guidelines for protection of aquatic life (CCME 1999c; Kelly et al., 2010). However, in another study, Hg and MeHg in snowmelt are below the Canadian water quality guidelines for the protection of aquatic life (CCME 1999c; Kirk et al., 2014). Hg concentrations in lake sediments are below Canadian sediment quality guidelines for the protection of aquatic life except in one lake (CCME, 1999a; Neville et al., 2014).

Hg concentrations in fish collected in the Athabasca River near oil sands development decreased from 1984 to 2011. In Lake Athabasca, Hg concentrations in fish decreased or exhibited no trend. Hg concentrations in fish from Nemur Lake near oil sands mining areas increased from 2000 to 2007; however, the increase in concentrations is similar to that at remote lakes elsewhere (Evans and Talbot, 2012). Hg has been detected in waterbird eggs in the Athabasca River downstream of oil sands development and at a remote site on the Peace River. There has been a 40% increase in Hg burden in eggs from 1977 to 2009. However, the egg Hg concentrations are below the concentrations that would lead to reproductive damage. The effects of Hg exposure also depend on the type of waterbirds (Hebert et al., 2011). Hg concentrations in waterbird eggs collected downstream of oil sands development have increased compared to the year of earliest collection. The study suggests that the oil sands

development or local sources of Hg in the AOSR are impacting egg Hg levels or there are other factors conducive to MeHg accumulation; however, it is unrelated to forest fire events and long range transport of Hg. Some egg samples exceed the lower limit of the threshold for reproductive impairment (Hebert et al., 2013). Hg in wetlands waters are below Canadian water quality guidelines for protection of aquatic life. Hg concentrations in water and wood frogs varied spatially; however, the concentrations are not related to distance from bitumen upgraders (Akhter et al., 2015).

Overall, most of the studies show that Hg concentrations in water or sediments are below the concentrations that would result in wildlife health effects. In addition, Hg levels in wildlife are low and provide little evidence (for waterbird eggs) or no evidence (for fish) of deleterious effects. However, the different trends in Hg levels reported for different wildlife species require further study.

Trace elements including Cd, Cu, Pb, Ni, Ag, and Zn in snowmelt and water near oil sands development or downstream of the development have exceeded Canada and Alberta water quality guidelines for protection of aquatic life (CCME 1999a; Kelly et al., 2010), but it is unclear if these exceedances were driven by deposition or non-deposition (e.g., runoff, leaching) processes. The exceedances in snowmelt were also observed for Pb, Zn, Fe and Al in another study; however, the impact of trace elements in snowmelt on the water quality of lakes and rivers remains unclear (Kirk et al., 2014). It is important to note that Canadian water, soil or sediment quality guidelines are not available for many of the trace elements, which poses a challenge in assessing the potential ecological effects.

Some of the trace element concentrations at oil sands sites exceed Canadian soil quality guidelines for the protection of environmental and human health (CCME, 1999b), but only some elements are vital to plant growth while other elements could be toxic to plants. Element uptake by plants varied throughout the AOSR, but is not considered elevated and no phytotoxic effects are found in plants. A higher uptake of elements are found in plants located in undisturbed sites likely due to the lower soil pH compared to the more alkaline soil at oil sands sites (Boutin and Carpenter, 2017). In another study near the Cold Lake area, metal contents in the soil are below the Canadian soil quality guidelines (CCME 1999b; Skierszkan et al., 2013). Arsenic is the only trace metal with sediment concentrations in the Cold Lake area exceeding the Canadian sediment quality guidelines for protection of aquatic life (CCME 1999a; Skierszkan et al., 2013). Higher deposition of mineral dust in the AOSR could impact vegetation in bogs. Since the bogs in the AOSR are naturally acidic, increased input of minerals to the bogs increase the alkalinity of the soil, which may be less or more favourable to other types of vegetation (Mullan-Boudreau et al., 2017).

Trace element concentrations have been analyzed in terrestrial sentinel animals, such as deer mice and meadow vole, at a reclaimed site near oil sands development and a reference site. Al, Ba, Cd, Hg and Sr concentrations in the animal kidneys are lower at the oil sands site than the reference site, suggesting other sources of contamination at the reference site. However priority pollutants, e.g. Co, Se, and Tl, in the mice kidneys were higher at the oil sands site than the reference site. Individually, these elements are potential teratogens and carcinogens (Rodríguez-Estival and Smits, 2016).

Despite exceedances of Canadian water, soil and sediment quality guidelines periodically at some sites, the resulting health impacts to wildlife in the AOSR remain unclear. The high levels of dust from oil sands extraction is a significant source of trace elements to ecosystems via atmospheric deposition; however, the ecological effects for some trace elements (e.g. V, Ni, and Mo) are unknown due to the absence of Canadian guidelines. Furthermore, there have been very few studies on trace element concentrations or effects on animals.

4.2 Significant Knowledge Gaps

Uncertainties in Trace Element Sources – Based on the NPRI, trace elements are disposed of in larger quantities in tailings than other forms of releases; however, it is not known whether these pollutants are released into air, soil and/or nearby waters. Studies identified fugitive dust as a major source of trace elements in the AOSR, whereas trace element emissions from fugitive dust and non-point sources in the NPRI are low. The differences in trace element emissions reported need to be reconciled. Biomass burning, home heating and land clearing burning activities are also potential trace element sources in the AOSR (e.g., Cd largely comes from biomass burning). Source apportionment analysis can be conducted using existing trace elements data to identify and quantify sources in the AOSR. A spatially-resolved emissions inventory for trace elements is lacking for modeling its transport and deposition.

Fugitive Dust Emissions – Hg and trace elements in deposition are predominantly from fugitive dust since they are typically bound to particles. Chemical fingerprinting methodologies can be applied to dust samples to determine the chemical characteristics or signatures. Dust emissions can be monitored using passive samplers or turf/surrogate surfaces which capture all particle sizes.

Fugitive Dust Deposition - The deposition rates of particles vary with size. MOUDI impactors can be deployed to determine the size distributions of particulate Hg and trace elements to gain a better understanding of the spatial patterns in the deposition of dust. Currently, there are no such measurements in the AOSR. The data will also inform Hg deposition modeling which typically exclude the Hg in the coarse fraction. Fugitive dust deposition is highly wind dependent; episodic wind storms can result in extremely high deposition during some times of the year. Instead of pollutant-specific deposition models, simple modeling of the meteorology, particle trajectories and dust deposition at different particle sizes and wind speeds are also recommended to obtain a general understanding of the fate of fugitive dust. In general, there is a lack of deposition modeling studies for trace elements which could provide a greater understanding on the contribution of various sources to deposition and identify specific areas with high deposition fluxes to inform ecosystem risk assessments.

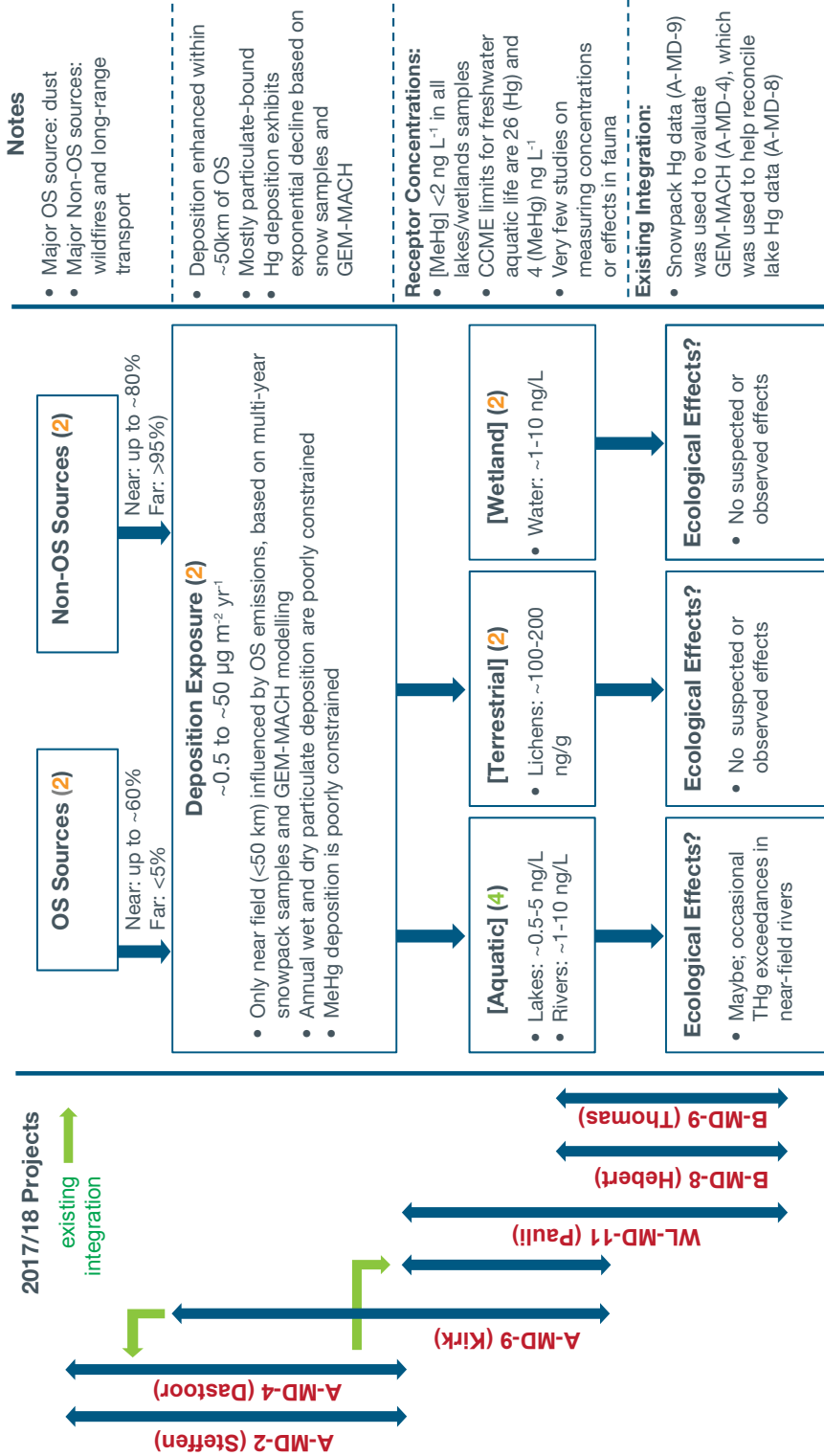
Hg and Trace Element Wet Deposition Monitoring – The wet deposition flux of Hg and trace elements throughout the year is another knowledge gap. There are plans to collect Hg wet deposition at AMS13 which is co-located with snowpack sampling; another site is needed upwind (e.g. AMS21-Stony Mountain) to establish background Hg wet deposition rates. Precipitation concentrations of trace elements are also needed to determine the wet deposition of trace elements; the data can also be used for the development and evaluation of wet deposition models.

Exposure of Hg and Trace Elements to Sentinel Biota – There are limited studies on the exposure of Hg and trace elements in sentinel animals in the AOSR; such studies are potentially useful in assessing the extent of trace element exposure and health impacts in wildlife. Many studies are comparing receptor concentrations in the AOSR to applicable Canadian environmental quality guidelines to assess potential ecological effects; however, the Canadian guidelines for some trace elements are not available. The fugitive dust generated from open-pit mining, vehicular traffic and petcoke will deposit trace elements to soil and aquatic environments. Water, soil and sediment quality guidelines need to be developed for all elements to gain a complete understanding of the potential ecological effects.

Ecological Risk Assessments – Maps of annual deposition loadings are needed to inform ecological risk assessments. Multi-media modelling has not been carried in the AOSR and may be useful for interpreting the observed/suspected effects, filling in the knowledge gaps in the different environmental compartments, and informing monitoring. Many challenges remain in assessing risks to ecosystems because of the exposure of biota to complex contaminant mixtures. There are already uncertainties on the toxicities of individual pollutants; the combined health effects from different pollutants are unknown.

4.3 Summary Figures

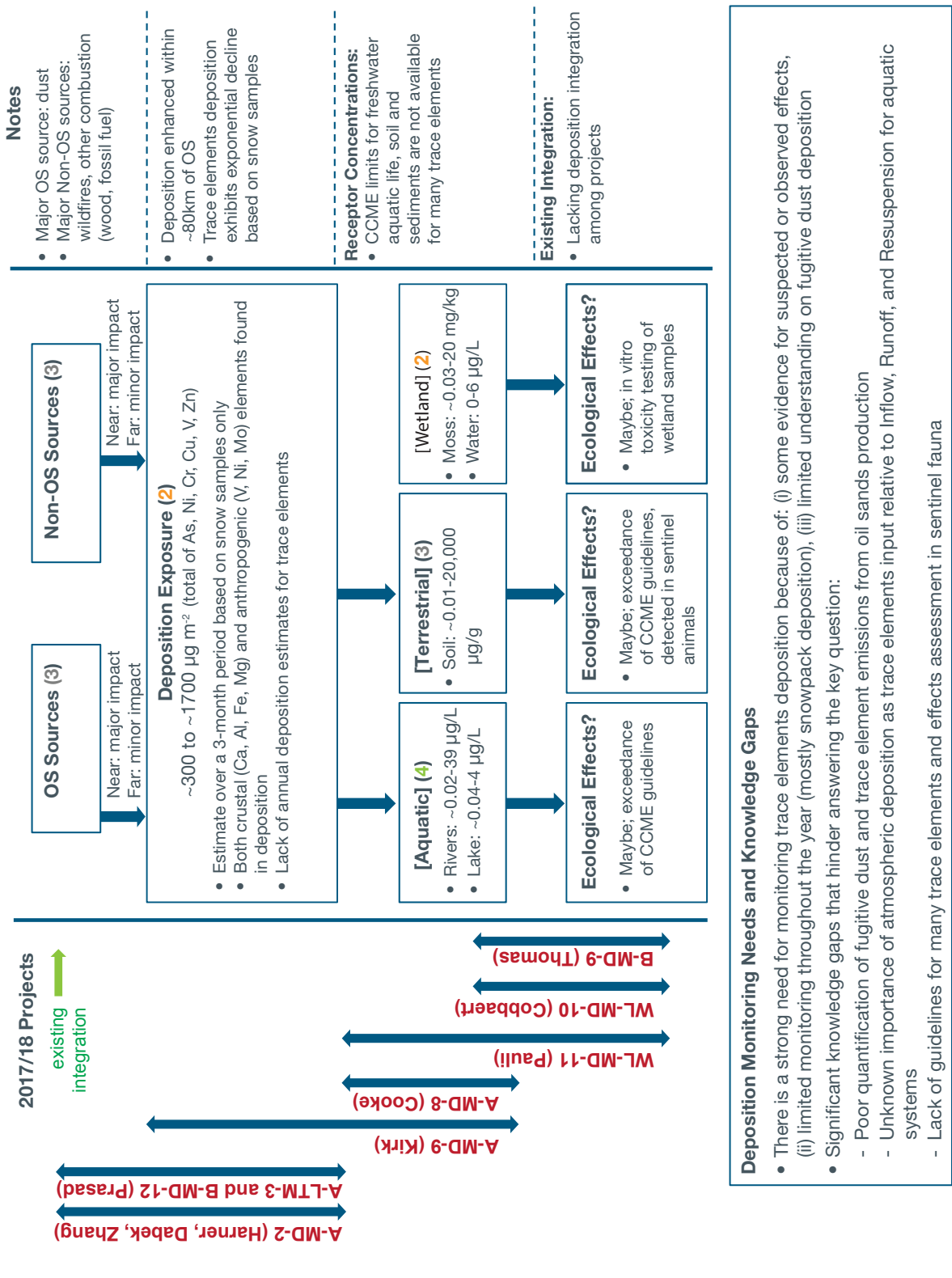
Could atmospheric Hg deposition from OS emissions be affecting ecosystems? Maybe



Deposition Monitoring Needs and Knowledge Gaps

- There is a strong need for monitoring PAC deposition because of: (i) some evidence for suspected or observed effects, (ii) limited monitoring throughout the year (mostly snowpack deposition), (iii) limited understanding on fugitive dust deposition
- Significant knowledge gaps that hinder answering the key question:
 - Poor quantification of PAC emissions from oil sands production
 - Unknown importance of atmospheric deposition as PAC input relative to Inflow, Runoff, and Resuspension for aquatic systems
 - Lack of guidelines for many PACs and effects assessment in sentinel fauna

Could atmospheric trace elements deposition from OS emissions be affecting ecosystems? Maybe



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5.0 Polycyclic Aromatic Compounds (PACs)

5.1 High-Level Overview of Current Knowledge

This section only provides a brief overview of the work-to-date on understanding the sources, transport, deposition, fate, and effects of PACs in the Oil Sands Region. An extensive synthesis report has been prepared as part of another OSM project (R-1-1718 Air Evaluation Integration Synthesis and Reporting) and has been submitted to a journal for publication as a review paper (Harner et al., 2018). This parallel effort has bolstered integration and identified priority areas for future PACs monitoring.

5.1.1. Sources

Polycyclic aromatic compounds (PAC) are made up of parent polycyclic aromatic hydrocarbons (PAH), alkylated PAHs, parent and alkylated dibenzothiophenes (DBT), and heterocyclic aromatic compounds (Boström et al., 2002; Schuster et al., 2015; Manzano et al., 2017). In the AOSR, PACs are emitted from oil sands development, such as bitumen production facilities, open-pit mining, mine fleet, haul roads, petcoke dust, and tailings ponds, as well as from non-industrial sources like wood burning, forest fires, and vehicular emissions (Jautzy et al., 2013; Galarneau et al., 2014; Parajulee and Wania, 2014; Hsu et al., 2015; Zhang et al., 2016; Qiu et al., 2018). While parent PAHs are emitted from both natural and anthropogenic sources, alkylated PAHs and DBTs and heterocyclic aromatic compounds are predominantly emitted from petrogenic sources (Kelly et al., 2009; Jautzy et al., 2013; Wickliffe et al., 2014; Schuster et al., 2015; Manzano et al., 2017). Several studies suggest that wind-blown dust from petcoke stockpiles and fugitive dust from the mining areas are major sources of PACs (Jautzy et al., 2013; Zhang et al., 2016). Other studies suggest a significant proportion of PAHs are volatilized from tailings ponds (Galarneau et al., 2014; Parajulee and Wania, 2014). The use of molecular diagnostic ratios has identified petroleum combustion and other types of combustion as sources of PACs to ambient air, deposition and sediment (Jautzy et al., 2013; Cho et al., 2014; Schuster et al., 2015; Manzano et al., 2016).

The annual emissions of speciated PAHs (parent PAHs and several nitrogen-containing PAHs) from oil sands facilities are reported to the NPRI (ECCC 2017). The PAH emissions are apportioned into stacks, fugitive, non-point sources, direct discharge, storage/handling, and spills. In 2015, the largest PAH emissions were from point sources followed by fugitive and storage/handling releases. The amount of PAHs disposed in tailings is significantly larger than point source emissions, but the amount of PAHs volatilized from tailing ponds is not quantified in the NPRI. It is estimated that PAH fluxes from tailings ponds are 4.6 times of the point source and fugitive emissions reported in the NPRI in 2012 (Galarneau et al., 2014).

Overall, the location and magnitude of speciated PAC sources are somewhat understood. Although the major source sectors for PACs have been identified, the quantification of speciated PAC emission rates remains difficult due to the complex nature of PACs and analytical challenges (Qiu et al., 2018).

5.1.2. Deposition

PAC deposition measurements and modeling have been conducted in the AOSR. Several studies measured PAC deposition to snowpacks during a 3-4 month period (Kelly et al., 2009; Bari et al., 2014; Cho et al., 2014; Manzano et al., 2016). The studies found that PAC deposition is higher near major oil sands development than more distant sites, and decline exponentially with distance. This is because most of the PACs are bound to particles that have a tendency to deposit near its emission source. At the site of major oil sands development, the maximum PAC deposition to snowpacks can range from 1000 to 7870 $\mu\text{g}/\text{m}^2$ over a 3-4 month period. Within 50 km of major oil sands development, the total PAC deposited into snowpacks can range from 392 to 1800 kg over a 3-4 month period. Alkylated PACs are the dominant PAC species in the snowpacks in two studies (Bari et al., 2014; Manzano et al., 2016), whereas DBTs, phenanthrene/anthracene, fluoranthene/pyrene, chrysene, and fluorene are the most abundant PAC species in snowpacks sampled in another study (Cho et al., 2014). In terms of the spatial patterns, the highest PAC deposition to snowpacks is observed over the Athabasca River between the Muskeg and Steepbank Rivers where the oil sands development is most intense (Manzano et al., 2016). Higher deposition is also found along the north-south directions than east-west directions, since oil sands developments are concentrated along the Athabasca River which runs in the north-south direction. Air mass transport also influences the spatial patterns of the PAC deposition because trajectories traveled farther distances in the north-south directions than east-west directions (Cho et al., 2014).

The dry deposition of PACs over different surfaces (e.g., water, forest, grass and shrubs) has been estimated using measured air concentrations and modeled dry deposition velocities (Zhang et al., 2015b). Depending on the surface and PAC species, the annual dry deposition flux can range from 170 to 5380 $\mu\text{g}/\text{m}^2$ at sites near oil sands development. Dry deposition contributes more to the total deposition of PACs than wet deposition. Consistent with PAC deposition measurements in snowpacks, a significant portion of the dry (80%) and wet (60%) deposition fluxes is attributed to alkylated PAHs. The wet scavenging efficiency of PACs by rain and snow has been examined using a scavenging ratio (W) method (Zhang et al., 2015a). W is the ratio of pollutant concentration in precipitation to that in ambient air; this parameter could potentially be used to estimate wet deposition at AOSR locations where only the air concentrations are monitored. Scavenging by snow is estimated to be 10 times more efficient than scavenging by rain for gas-phase and particulate-phase PACs. Scavenging of particulate-phase PACs is estimated to be 5 to 10 times more efficient than gas-phase PACs. It suggests that the scavenging of particulate-phase PACs by snow is a major contributor to the total wet deposition of PACs in the AOSR. These findings corroborate the PAC deposition measurements in snowpacks.

Overall, total PAC deposition is well understood. Measurements of wet deposition and air concentrations have been used to develop deposition models, which are in general agreement with snowpack measurements.

5.1.3. Receptor Concentrations

PACs have been measured in air, water, soil, sediments, tree cores and lichens in the AOSR. In ambient air, the measurements include the aforementioned PACs and novel heterocyclic aromatic compounds. Using passive or active air sampling methods, alkylated PAHs are found to be more abundant than DBTs and parent PAHs in ambient air (Harner et al., 2013; Schuster et al., 2015; Zhang et al., 2015a). Similar to snowpack deposition, PAC air concentrations decline exponentially with distance from major oil sands production. Alkylated PAHs and DBTs exhibit a more rapid decline than parent PAHs because they tend to be bound to particles that deposit rapidly. In contrast, parent PAHs show a weaker concentration gradient with distance from major oil sands development. This is because many of the parent PAHs are in the gas phase, which can undergo long-range transport. Forest fires and volatilization of previously-deposited PACs from soil and lakes are other potential sources of parent PAHs at distant sites (Harner et al., 2013; Hsu et al., 2015; Schuster et al., 2015). Heterocyclic aromatic compounds that contain nitrogen or sulfur have also been found in air samples and in petcoke with similar speciation profiles, which suggest the impact of petcoke on PAC concentrations downwind. These novel compounds may serve as pollutant markers of oil sands activities given the lack of natural emission sources of heterocyclic aromatic compounds (Manzano et al., 2017). PAH oxidation products, such as quinones, are present in ambient air near oil sands development and likely formed by secondary chemical reactions involving parent PAHs instead of direct air emissions (Wnorowski and Charland, 2017). Quinone concentrations can exceed its analogous parent PAH concentrations in the particulate phase with increased sampling times (Wnorowski, 2017).

PAC concentrations in aquatic ecosystems are somewhat understood. PAC concentrations in the Athabasca tributaries, Athabasca River and Lake Athabasca have been measured in one study (Kelly et al., 2009), and are routinely monitored by AEP. Among these locations, PAC concentrations are highest at the tributary mouth, which is the area most affected by land disturbances. These land disturbances cause concentrations to be 10 to 50 times higher than surface water concentrations at background sites and exceed the threshold that signifies toxicity to fish. Alkylated PAHs are the most abundant group of PACs in surface water. In lake sediments, several studies observed increasing trends in PAC concentrations in sediments (Timoney and Lee, 2011; Kurek et al., 2013; Evans et al., 2016). PAC levels in sediment cores from five lakes within 35 km of major oil sands development have increased by 2.5 to 23 times since the 1960s, which corresponds to the beginning of oil sands development. This is opposite in trend to PAC concentrations in sediment cores sampled in remote lakes, which peaked in the mid-1900s and have since declined. Alkylated PAHs make up a larger proportion than parent PAHs in the sediment (Kurek et al., 2013). An increasing temporal trend in PAH levels in sediments is also found near oil sands development, whereas weaker temporal trends are observed in the Athabasca Delta and Lake Athabasca which are sites far away from oil sands development (Evans et al., 2016). Heterocyclic aromatic compounds have also been detected in lake sediments with concentrations declining with distance from major oil sands development (Manzano et al., 2017).

PAC concentrations in terrestrial ecosystems are somewhat understood. In soil, parent and alkylated PAH concentrations are higher at sites near oil sands development and tend to decrease with distance (Boutin and Carpenter, 2017). In the Cold Lake oil fields (~300 km from surface mining region), parent and alkylated PAH concentrations measured in most of the soil samples are low (Korosi et al., 2013). Alkylated PAHs in soil are more abundant than parent PAHs (Korosi et al., 2013; Boutin and Carpenter 2017). Higher alkylated PAHs in soil may be due to contamination by effluent or land disturbance instead of atmospheric deposition (Korosi et al., 2013). Concentrations of phenanthrene, pyrene and benzo[a]pyrene in soil have also been predicted using a multi-media model (Parajulee and Wania, 2014). The predicted concentrations depend on the emissions scenario considered in the model. The emissions scenario that includes both direct air emissions and evaporative emissions from tailings ponds resulted in modeled concentrations that are closer to measured soil concentrations compared to the emissions scenario with only direct air emissions which underestimated the observed soil concentrations. In lichens, alkylated PAH and DBT concentrations can be up to 4 times the parent PAH concentrations. The concentrations in lichens are higher near oil sands operations and lower at distant sites (Studabaker et al., 2017).

PAC concentrations in wetland ecosystems are poorly understood. The only such study found that the concentration of PACs collected in semi-permeable membrane devices (SPMD) decreases exponentially with distance from bitumen upgraders and are dominated by alkylated PAHs and DBTs (Pauli and Mundy, 2017).

5.1.4. Ecological Effects

Parent PAH concentrations in soil are below the Canadian soil quality guidelines in one study (CCME 1999; Korosi et al., 2013); however in another study, soil concentrations near oil sands development exceed the Province of Ontario and USEPA guidelines (Boutin and Carpenter, 2017). In sediment, some of the parent PAH species exceed the Canadian sediment quality guidelines (CCME 1999; Kurek et al., 2013; Evans et al., 2016). In a long-term study, five parent PAH species exceeded the sediment guidelines for almost two decades (CCME 1999; Kurek et al., 2013). The comparison of AOSR receptor concentrations with Canadian environmental quality guidelines is limited to parent PAHs. Guidelines for alkylated PAHs and DBTs have not been developed due to a lack of toxicology data.

PAH concentrations in plants are higher near oil sands development than at sites east or west of the oil sands development. The concentrations are also higher at sites with elevated soil PAH concentrations (Boutin and Carpenter 2017).

PAHs are capable of causing carcinogenic, teratogenic and genotoxic effects in laboratory animals and in humans exposed to PAHs in occupational settings (Kim et al., 2013; Wickliffe et al., 2014). Analysis of a sentinel zooplankton (*Daphnia*) in lakes indicates no harmful effects on the *Daphnia* populations and increasing abundances in some instances. The latter may be attributed to the warming climate, which increases algal production in lakes (Kurek et al., 2013). PACs have been detected in tadpoles, recent metamorphs, and adult wood frogs in wetlands (Pauli and Mundy 2017). Concentrations varied between sites; however, there is no relationship with distance to bitumen upgraders. Using *in vitro* toxicity testing, the greatest risk of toxicity

to PACs is found near bitumen upgraders (Pauli and Mundy 2017). In waterbird eggs, PAH concentrations are low except for slightly higher concentrations near the receiving waters of the Athabasca River (Hebert et al., 2011). Large mammals in the AOSR including moose, wolves and woodland caribou have been exposed to PACs. The abundance of alkylated PAHs in moose and wolf scat samples indicates the exposure was primarily from petrogenic sources (Lundin et al., 2015). PAH concentrations have been measured in different types of fish collected in the Athabasca and Slave Rivers (Ohiozebau et al., 2017). The concentrations in fish are higher in the Athabasca River near oil sands development than downstream locations in the Slave River. The PAH levels in fish are not expected to pose a cancer risk to the people consuming it (Ohiozebau et al., 2017). In an earlier study, surface water concentrations exceeded the threshold for fish toxicity (Kelly et al., 2009). A potential link has been observed between increased alk-PAH exposure at sites near oil sands development and endocrine disruption in river otters (Thomas et al., 2017).

The toxicity of ambient air samples in the AOSR has been evaluated using in vitro mutagenicity and cytotoxicity tests (Jariyasopit et al., 2016). Although not technically deposition, inhalation of these air samples pose a weak mutagenicity risk. The highest mutagenicity is observed near major oil sands development, and the mutagenicity levels increase with PAC and oxygenated PAC concentrations. Cytotoxic potential is found in the air samples at varying levels across different sites and do not correlate with PAC concentrations.

5.2 Significant Knowledge Gaps

Methodologies for monitoring and quantifying PACs – There are large differences in the analytical methods and standard operating procedures that make it difficult to compare or combine the data obtained using the different methods. Standard reference materials (SRM) should be used in all labs to validate the analytical methods. SOPs do not necessarily have to be standardized across all labs because of ongoing updates and improvements. It is important to standardize sampling protocols and ensure that the sampling and deployment of devices are done correctly.

Emission Sources and Speciation – Natural and anthropogenic source contributions to parent PAHs in the AOSR have not been quantified. In the NPRI, oil sands facility emissions (point sources and fugitive sources) are reported mainly for parent PAHs; however, alkylated PAHs, DBTs and other heterocyclic aromatic compound are often present in higher concentrations in ambient air, soil, water, and sediment samples than parent PAHs. The emissions inventory needs to account for other PACs and different sources in order to assess how anthropogenic emissions are affecting ecological systems in the oil sands region.

Fugitive Dust Emissions – A portion of PACs in deposition is likely from fugitive dust. Chemical fingerprinting methodologies can be applied to dust samples to determine the chemical characteristics or signatures. Dust emissions can be monitored using passive samplers or turf/surrogate surfaces that capture all particle sizes.

Fugitive Dust Deposition - The deposition rates of particles vary with the size of particles. MOUDI impactors can be deployed to determine the size distributions of PACs to gain a better understanding of the spatial patterns in the deposition of dust. Currently, there are no such measurements in the AOSR. Fugitive dust deposition is highly wind dependent; episodic wind storms can result in extremely high deposition during some times of the year. Instead of pollutant-specific deposition models, simple modeling of the meteorology, particle trajectories and dust deposition at different particle sizes and wind speeds are also recommended to obtain a general understanding of the fate of fugitive dust.

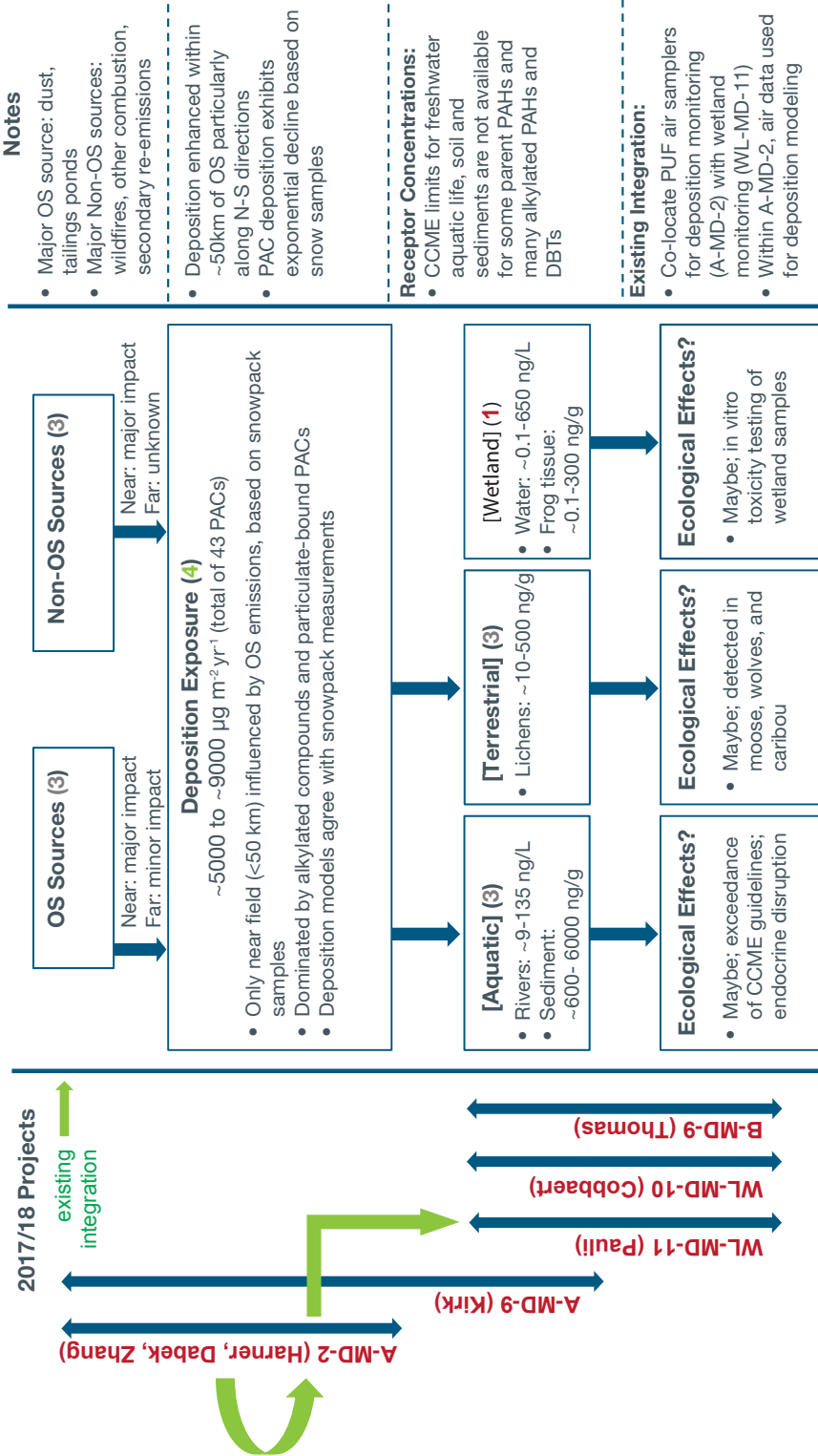
Deposition-Receptor Relationships – The deposition of PACs in the AOSR has been quantified; however, it has not been explicitly linked to receptor or wildlife concentrations. The use of predictive models (air quality, fate and transport models) may improve the understanding of sources, transformation and deposition processes impacting receptor concentrations of PACs and post-deposition impacts in the AOSR.

Ecological Effects – Many studies are comparing receptor concentrations in the AOSR to applicable Canadian environmental quality guidelines to assess potential ecological effects; however, the Canadian guidelines for alkylated PAHs, DBTs, heterocyclic aromatic compounds, and PAC transformation products are not available. These compounds are predominantly derived from petrogenic sources and often are present in higher concentrations in ambient air, soil, water, and sediment samples than parent PAHs.

Ecological Risk Assessments – Maps of annual deposition loadings are needed to inform ecological risk assessments. Multi-media modelling has not been carried in the AOSR and may be useful for interpreting the observed/suspected effects, filling in the knowledge gaps in the different environmental compartments, and informing monitoring. Many challenges remain in assessing risks to ecosystems because of the exposure of biota to complex mixtures. There are already uncertainties on the toxicities of individual pollutants; the combined health effects from different pollutants are unknown.

5.3 Summary Figure

Could atmospheric PAC deposition from OS emissions be affecting ecosystems? Maybe



Deposition Monitoring Needs and Knowledge Gaps

- There is a strong need for monitoring PAC deposition because of: (i) some evidence for suspected or observed effects, (ii) limited monitoring throughout the year (mostly snowpack deposition), (iii) limited understanding on fugitive dust deposition
- Significant knowledge gaps that hinder answering the key question:
 - Poor quantification of PAC emissions from oil sands production
 - Unknown importance of atmospheric deposition as PAC input relative to Inflow, Runoff, and Resuspension for aquatic systems
 - Lack of guidelines for many PACs and effects assessment in sentinel fauna

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6.0 Recommendations for Integration and Additional Monitoring

This section provides recommendations to begin addressing significant knowledge gaps identified in sections 3.2-5.2 by suggesting integration between existing projects and/or additional monitoring. These recommendations are intended to serve as a next step that will inform subsequent FS and LTM planning, as opposed to completely filling the gaps. Additional monitoring is only suggested here on a limited spatial and temporal scale, as its intention is to ascertain what scale of monitoring (if any) is needed. In other words, it is possible that integration and additional monitoring may reveal that a presumed knowledge gap is not actually significant or is adequately addressed by current monitoring.

6.1 Acidifying and Eutrophying Pollutants

Significant Gap: Interpreting IER Data

Recommendation #1: hold a workshop for IER experts and PIs on projects that use IERs (A-LTM-3 and WL-MD-10). Outcomes of the workshop should include: (i) agreement on and use of single SOP, (ii) co-ordination of deployment/analysis, (iii) decide whether to consolidate all IER measurements into a single OSM project, and (iv) plan a co-location study (see recommendation #2).

Recommendation #2: co-locate IER measurements at several sites with wet only and total deposition measurements of N, S and base cations for 3 years (A-MD-2 Enhanced Deposition). A more in-depth comparison of IER measurements versus wet/ total deposition will give insight on how to properly interpret IER data.

Significant Gap: Base Cation Deposition

Recommendation #3: integrate existing datasets of base cation atmospheric concentrations and base cation deposition (A-MD-2 Enhanced Deposition, A-LTM-3 Deposition to Forests, A-MD-4 Modelling, WL-MD-10 Wetland Monitoring) to ascertain if the following are consistent between datasets: (i) spatial patterns, and (ii) relative abundances of base cation species. This exercise would help evaluate the GEM-MACH model, and hence provide an assessment on our current understanding of base cation sources. If datasets are generally consistent then GEM-MACH can be used to fill in monitoring gaps (e.g., dry deposition). However, if datasets are disparate then additional monitoring will likely be needed.

Recommendation #4: conduct speciated PM measurements at finer size resolution at several enhanced deposition sites for 1-3 years. This will provide detailed information on size-resolved base cation composition and allow for a much better estimate of base cation dry deposition. Results can be used to evaluate the efficacy of current methodologies (SASS, IER, filter packs) and improve models (GEM-MACH). The monitoring could be conducted with a Micro-Orifice Uniform Deposit Impactor (MOUDI). Care would need to be taken to ensure measurements are not influenced by local sources (e.g., nearby roads)

Significant Gap: NH₃ Sources and Bi-directional Exchange

Recommendation #5: integrate existing surface [NH₄⁺] and pH measurements to determine which ecosystem components could be emitting NH₃ (WL-MD-10 Wetland LTM, B-MD-12 Forest Health Monitoring, WL-MD-11 Amphibian and Wetland Health). Current emission potential ($f = [\text{NH}_4^+]_{\text{surface}} / [\text{H}^+]_{\text{surface}}$) parameterizations in GEM-MACH are crucial for simulating NH₃ surface-air exchange but are not AOSR specific. Comparing measured and modelled f could greatly improve bi-directional exchange in GEM-MACH and would determine which ecosystems are net emitters of NH₃. If ecosystems could be strong sources and this affects the N-deposition budget, then an ammonia flux measurement FS might be warranted. However, an NH₃ flux measurement study would be technically challenging.

Significant Gap: Total N Deposition

Recommendation #6: integrate existing measurements of NO_y monitoring (A-MD-2 Enhanced Deposition, WBEA Air Quality Network) and available literature to determine if oxidized N species not currently monitored (e.g., PAN, PPN, organic nitrates) could make up a significant fraction of NO_y. If a significant fraction of NO_y is not currently monitored species (i.e. other than NO, NO₂, HNO₃, pNO₃⁻), then additional monitoring may be needed to understand the impact on total N deposition. GEM-MACH modeling results could be used as a first estimation.

Recommendation #7: conduct focused study (1-3 years) at one or several enhanced deposition sites to measure Total N deposition and its components. The purpose is to assess the: (i) fraction of total N deposition currently being captured by the network, (ii) major N-species in air, and (iii) dominant deposition pathway for each major N pollutant. Ideally this study could be conducted at several ecological sites along an N-deposition gradient; however, this is likely not feasible due to site requirements (e.g., power, frequent site access). Furthermore, enhanced deposition sites already measure several key N-species.

Significant Gap: Regional N, S and Base Cation Deposition

Recommendation #8: this gap is being addressed by A-MD-2 (N and S Deposition) and A-MD-4 (Emissions, Transformation and Fate). The former is monitoring an extensive suite of N, S and base cation deposition at 2 sites in northern Saskatchewan (~350 km downwind; base cation measurements not supported by OSM), and the latter is conducting aircraft-based measurements in 2018 to assess N, S, and base cation emissions, transformations and deposition (up to 100s of km downwind). These studies may highlight the need for additional N, S and base cation monitoring at far field distances (>150 km).

Significant Gap: Disseminating Deposition Surfaces

Recommendation #9: develop a process to share modelled (e.g., GEM-MACH; A-MD-4) and interpolated (e.g., WL-MD-10, A-LTM-3) deposition surfaces (maps). This data exchange should be a two-way street, since monitoring data is necessary to evaluate and improve modelling efforts and interpolated deposition surfaces. Since this gap is related to a broader OSM challenge of effective and efficient data sharing, a comprehensive process is not suggested here. Instead, workshop participants will strive to contact relevant PIs for

opportunities to integrate deposition surfaces and/or monitoring data that will support their projects. For this ad hoc approach to work, there needs to be sufficient awareness amongst PIs of the available data products, as well as their utility and limitations.

6.2 Trace Elements and Hg

Significant Gap: Trace element sources and fugitive dust emissions

Recommendation #10: conduct source apportionment analysis using trace elements and develop gridded emissions for trace elements. Trace element measurements at various monitoring sites in the AOSR can be used in source apportionment models to quantify the contributions of various sources to trace element concentrations and track the impact of emissions over time. A spatially-resolved emissions inventory for trace elements is needed for modeling its transport and deposition.

Recommendation #11: monitor fugitive dust emissions and determine chemical characteristics. Dust emissions can be monitored using passive samplers or turf/surrogate surfaces that can capture all particle sizes. It is recommended to conduct dust emissions monitoring at larger open-pit mining operations with significant dust emissions. The dust emissions measurements are potentially useful for modeling its transport and deposition. The chemical characteristics or signatures of the dust need to be determined (e.g., chemical fingerprinting methods) for source apportionment analysis and emissions inventory development.

Significant Gap: Fugitive dust deposition

Recommendation #12: monitor fugitive dust deposition. The deposition rates of particles are dependent on the size of the particles. MOUDI impactors can be deployed to determine the size distributions of particulate Hg and trace elements to gain a better understanding of the spatial patterns in the deposition of dust. Currently, there are no such measurements in the AOSR and measurements could be co-located with passive sampling and other deposition monitoring. The data will also inform Hg deposition modeling which typically exclude Hg in the coarse fraction and is important to the development of dry deposition models and source apportionment analysis. Fugitive dust deposition is highly wind dependent; hence, wind storms can periodically result in extremely high deposition rates. Deposition monitoring should be conducted throughout the year and could be accomplished through a combination of snowpack sampling during winter and passive samplers.

Recommendation #13: model fugitive dust transport and deposition. Modeling of the local meteorology, particle trajectories and dust deposition as a function of particle sizes and wind speeds are recommended to obtain a general understanding of the fate of fugitive dust. The model output would be useful for modeling the transport and dry deposition of particulate Hg and trace elements.

Note: This report was updated in March 2019 to correct a factual inaccuracy in paragraph 3 related to monitoring dust emissions at open-pit mining operations.

Significant Gap: Wet deposition of mercury and trace elements

Recommendation #14: wet deposition monitoring. Wet deposition measurements are essential to the development and evaluation of wet deposition models, which is an important component of total deposition modeling. There are plans to collect Hg wet deposition at AMS13 which is co-located with snowpack sampling; another site is needed upwind (e.g., AMS21-Stony Mountain) to establish background Hg wet deposition rates. Precipitation measurements of trace elements are also needed to determine the wet deposition of trace elements; this data can be used to develop wet deposition models.

Significant Gap: Linking deposition results with ecological impacts

Recommendation #15: produce deposition maps to inform ecological risk assessments. Maps of annual deposition loadings are needed to inform ecological risk assessments. In addition to quantifying the deposition rates, it would be ideal to estimate the relative contributions of various sources (e.g., point sources, fugitive dust, and biomass burning) to deposition. This would help address the overarching question, “Could anthropogenic emissions and/or deposition be affecting ecological systems in the oil sands region?”

Recommendation #16: conduct multi-media modeling. Multi-media modeling have not been carried out in the AOSR and may be useful for interpreting the observed/suspected effects and filling in the knowledge gaps in the different environmental compartments and supplementing monitoring gaps.

Recommendation #17: attributing causes of ecological effects. Besides atmospheric deposition, ecological effects can be attributed to numerous causes including effluent discharge, soil erosion, surface runoff, tributary and groundwater flows, and resuspension from aquatic systems. Physical, chemical and biological variables can also affect pollutant uptake and toxicity in biota. While it is important to establish links between atmospheric pollutant deposition and ecological effects, the non-deposition causes mentioned above should also be carefully considered.

Significant Gap: Uncertainties in the ecological effects

Recommendation #18: improve understanding of ecological effects of pollutant exposure. Many challenges remain in assessing ecological risks because of the exposure of biota to complex contaminant mixtures. There are already uncertainties for the toxicities of individual pollutants and Canadian environmental quality guidelines are not available for many of the pollutants; the resultant health effects from different pollutants are unknown. Other challenges relate to determining the size and extent of the effects and the level of biotic response that are considered significant, the spatial distribution required to capture changes in the ecosystem, and monitoring methods with the capability to detect effects. Furthermore, there have been very few studies measuring pollutants and assessing effects in sentinel fauna. Considering that the emphasis is on suspected or observed effects, it is recommended to further improve the understanding of the ecological effects of pollutant exposure.

6.3 PACs

Several of the major knowledge gaps for mercury and trace metals in section 6.2 are also applicable to PACs. Refer to **Recommendations #11 to #13** to address fugitive dust emissions and deposition; **Recommendations #15 to #17** to address establishing linkages between deposition results and ecological effects; and **Recommendation #18** to address the uncertainties in ecological effects.

Significant Gap: Methodologies for monitoring and quantifying PACs

Recommendation #19: use of standard reference materials (SRM) to validate analytical methods and standardizing sampling protocols. Analytical methods and SOPs for PACs vary across labs, which make it difficult to compare or combine the data obtained using the different methods. SRMs should be used in all labs to validate the analytical methods. SOPs do not necessarily have to be standardized across all labs because of ongoing updates and improvements. It is important to standardize sampling protocols and ensure that the sampling and deployment of devices are done correctly.

Significant Gap: Incomplete PAC emissions and speciation

Recommendation #20: development of a comprehensive emissions database for PACs. ECC's NPRI does not provide emissions data for all PAC species that are currently monitored in the ambient environment and does not include all sources in the AOSR. It is recommended that a comprehensive emissions database be developed to assess impacts of OS emissions on receptor concentrations and model ambient air concentrations and deposition.

6.4 Towards a Unified Deposition Monitoring Program

This deposition integration project has raised awareness between PIs on the scope, scale, and data products of existing OSM-funded deposition projects. Discussion at the workshop highlighted significant knowledge gaps that have yet to be addressed by the deposition monitoring system. Sufficient awareness and the identification of gaps are essential first steps towards creating a unified OS deposition monitoring program. However, the prioritization of deposition monitoring needs (and hence an evaluation of which monitoring might be extraneous) is also necessary for creating a unified program.

The shift from a mix of 14 somewhat disjointed projects to a unified OS deposition monitoring program is very challenging. Such a unified program would help to ensure: (i) monitoring data needs are being met for assessing ecosystem effects, and (ii) clarity on the scope of deposition monitoring, which would help promote integration. One potential path forward would be to start by designing a core deposition LTM program, within a pre-defined budget, using the following approach:

1. **Determine deposition data needs and priorities for assessing effects in terrestrial, aquatic, and wetland ecosystems** (i.e., which pollutants are relevant? What monitoring timescales are relevant? What spatial scales are relevant?). Some of these questions have been implicitly or explicitly addressed by this current project.

2. **Evaluate existing FS and LTM deposition projects** (i.e., what data products are currently being provided? Which project components are fulfilling data needs? What are the significant gaps in the current system?). The existing monitoring and significant knowledge gaps have been summarized by this current project.
3. **Adjust the existing monitoring system as necessary to develop a core deposition LTM program that fulfills data needs and priorities.** This would likely involve keeping some monitoring components intact, while adjusting or eliminating others in order to make resources available to fill significant gaps that have already been identified (e.g., fugitive dust deposition monitoring). This would be a challenging step and was not covered by this current project – such a task would likely require objective, technical experts (i.e., scientists or technical managers not involved in any of the existing studies).
4. **Develop a framework to identify and trigger FS.** A core LTM program would not be able to address every question or data need that arises. There needs to be a framework process to elicit FS that clearly address a significant knowledge gap. This framework is needed if the intent is to shift away from the current proposal-based project planning system and towards a unified deposition monitoring program.

There are likely other approaches to achieve a unified OS deposition monitoring program, and the above is just one suggestion that centers around the development of a core LTM deposition program. This current (2017/18) project was able to raise awareness and begin building ad hoc linkages and integration – developing a unified deposition monitoring program will require additional work.

Appendices

Appendix A – Monitoring Site Maps

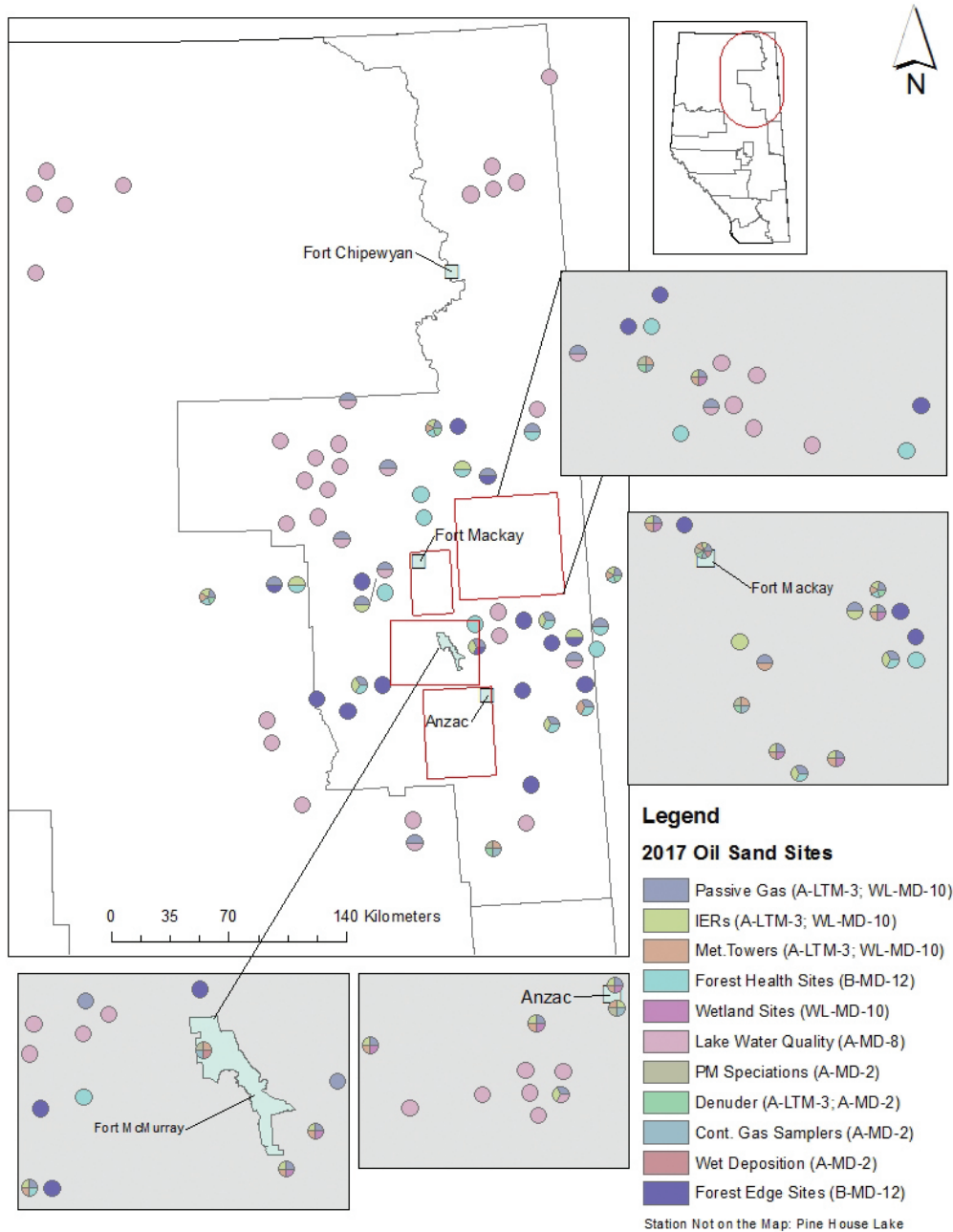


Figure A1. Acidifying and Eutrophying Pollutants

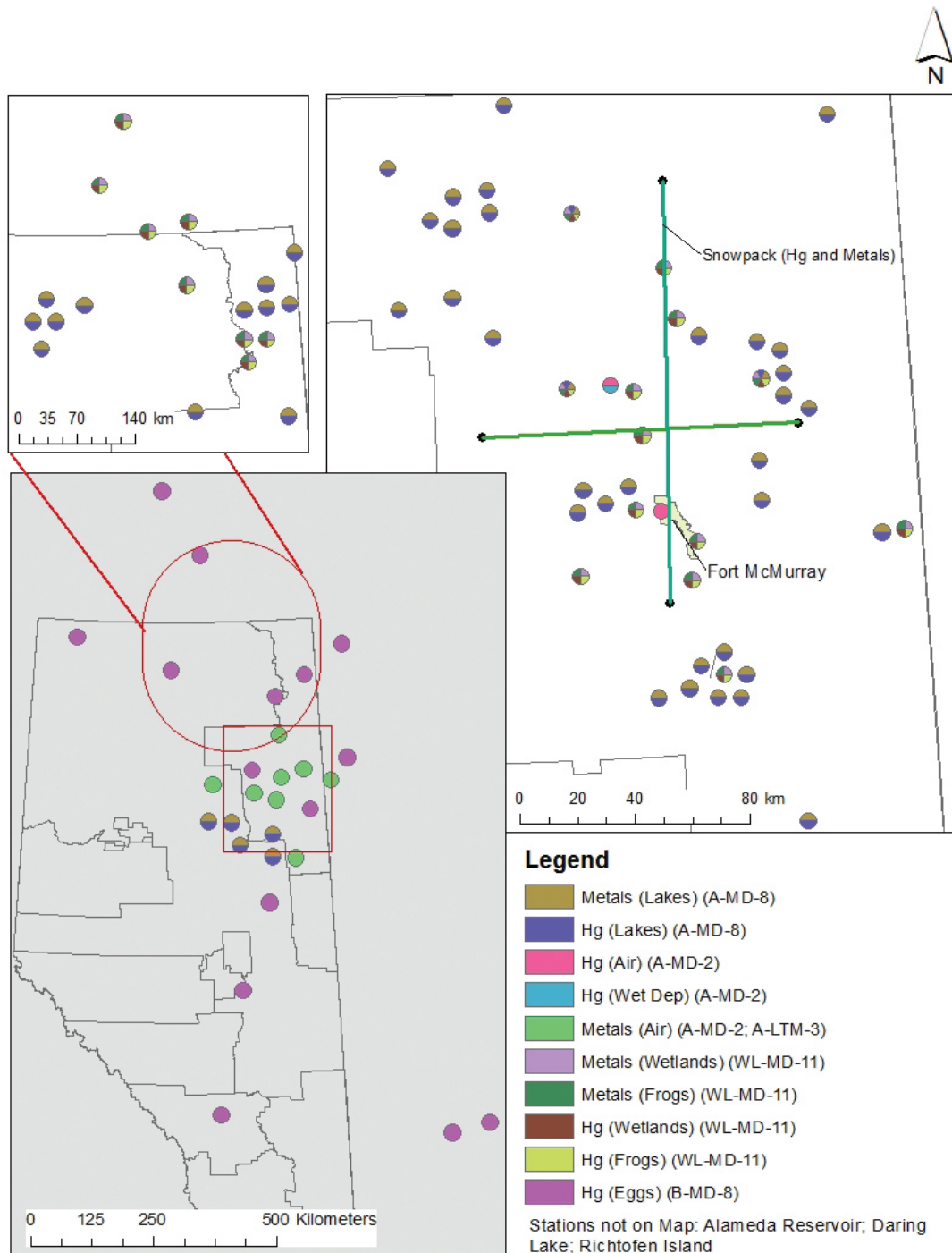


Figure A2. Hg and Trace Elements

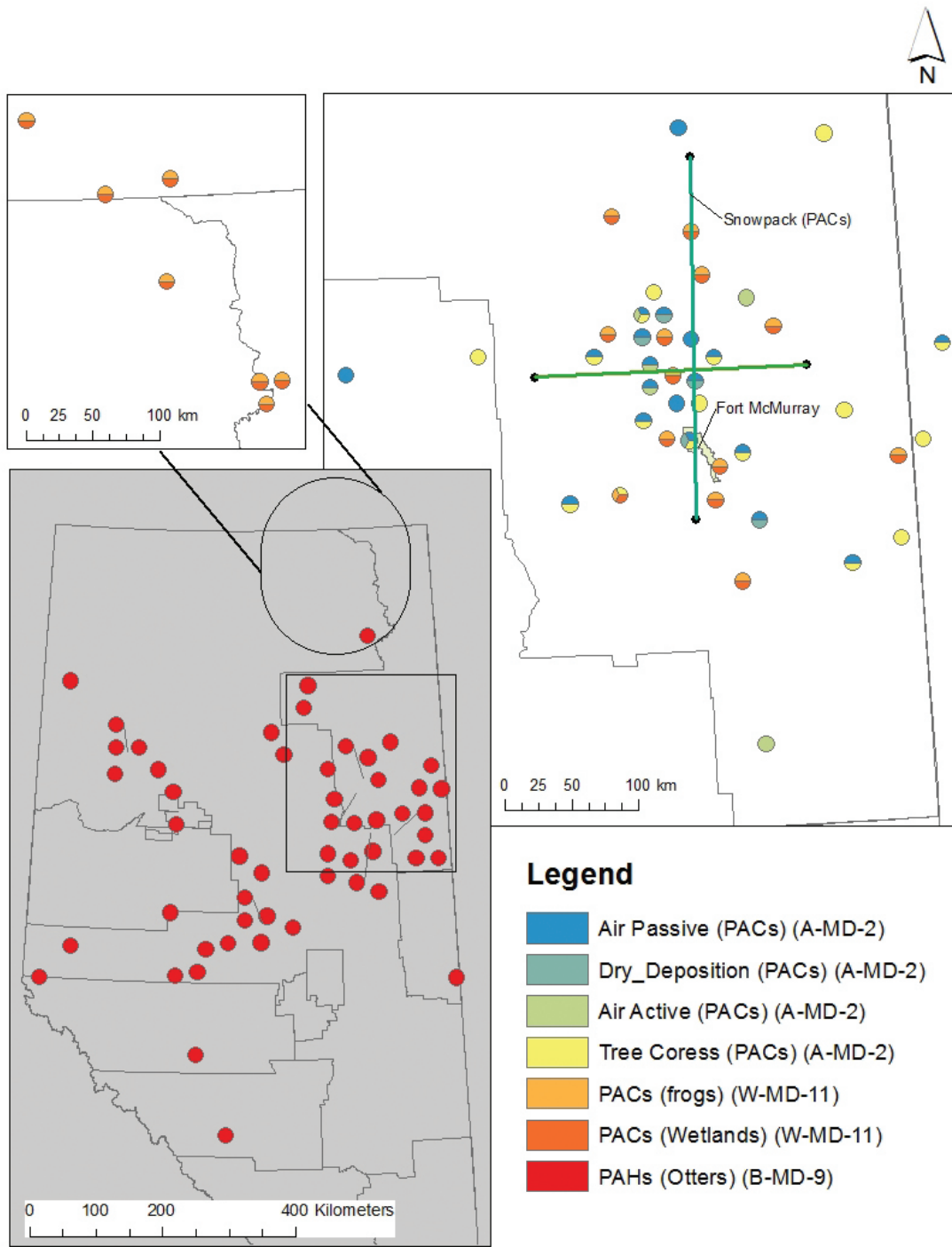


Figure A3. PACs

Appendix B - Project Summary Tables

Table B.1 Acidifying and Eutrophying Pollutants

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
A-MD-8 Acid Sensitive Lakes Monitoring	Shallow Lakes	pH	pH meter in situ	Annually in September; since 1995; grab samples	50 lakes extending outwards to ~300 km	Gibson et al., (2010a,b; 2015; 2016) Schmidt et al., (2010) Bennett et al., (2008) Andrews et al., (in prep)		Continue to monitor annually for compliance purposes
		Alkalinity	Gran Method					
A-LTM-3 Deposition Monitoring in Forests	Air	SO ₂ ²⁻ , Ca ²⁺ , K ⁺ , Mg ²⁺ , NO ₃ ⁻ , PO ₄ ³⁻	IC	(Bi)Monthly averages; since 2000; passive	30 forest sites extending to ~125 km	Passive Air Samplers Air concentration maps Dry deposition maps Hsu et al., (2016) Hsu (2013) Bytnerowicz et al., (2016) <i>Ion Exchange Resins</i> Bulk deposition maps Fenn et al., (2015) Watmough et al., (2014) Wieder et al., (2016a) Whitfield et al., (2009) <i>Modelling</i> Air concentration maps N, S and PAI deposition maps <i>Report</i> Clair and Percy (2015)	Explicitly linked with B-MD-12; provides deposition data to examine potential impacts from deposition on soil and vegetation parameters	Under review (may be replaced by denuders) Being evaluated at Air Monitoring Sites (may replace passive samplers) No longer used Continue to monitor with possibility of additional sites Continue to monitor
		NH ₄ ⁺ , NO ₃ ⁻ , TN	FIA					
		DIC, DOC, TDN, TN	TOC/TN Analyzer					
		HNO ₃ , NH ₃ , SO ₂ , NO ₂	Passive Air Samplers					
		HNO ₃ , NH ₃ , SO ₂ , PM _{2.5} (SO ₄ ²⁻ , NH ₄ ⁺ , NO ₃ ⁻ , trace elements)	Denuders					
		Deposition Velocity (N and S species)	CoTAG					
		Meteorology (T, RH, hv, wind speed/dir)	Meteorological Tower					
		SO ₂ ²⁻ , NO ₃ ⁻ , Ca ²⁺ , Mg ²⁺ , Na ⁺	Ion Exchange Resins					
		Ion exchange (NO ₃ ⁻ , H ₂ PO ₄ ⁻ , SO ₄ ²⁻ , NH ₄ ⁺ , K ⁺ , Ca ²⁺ , Mg ²⁺)	Plant Root Simulators					
		Deposition and air concentrations of: SO ₂ and NO _x	CALMET, CALPUFF, and CMAQ					
Deposition (Bulk)		Biannual averages; since 2007	21 forest sites extending to ~125 km					
Soil		Biannual averages; since 2011	49 forest sites extending to ~125 km					Under review (may be discontinued)
Modelling		N/A	Entire AOSR					No further modelling planned for this project

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring	
B-MD-12 Forest Health Monitoring	Soil	pH	pH meter (0.01 M CaCl ₂)	Every ~6 years; since 1998; grab sample	25 jack pine forest sites extending to ~125 km; 1998: 10 sites 2004: 12 sites 2011: 25 sites	Reports (and references therein) -AMEC (2000) -C.E. Jones et al., (2007) -Clair and Percy (2015)	Explicitly linked with A-LTM-3; deposition data is used to correlate receptor parameters with deposition in order to investigate causal relationships between deposition and forest health	Program review is underway for 2017/18	
		Total N, C, S	Dry Combustion Analyzer						
		NO ₃ ⁻ , NH ₄ ⁺	Colourmetric Flow Analyzer						
		SO ₄ ²⁻							
		Water-soluble ions	ICP-OES						
		Available P	Colourmetric Flow Analyzer						
		CEC	NH ₄ ⁺ replacement then ICP-OES						
		Base Cations:Al	Calculated						
		Biology (PLFA for microbial communities)							
		Tree density, crown closure, defoliation	Visual inspection						Every ~6 years; since 1998; grab sample
		Tree diameter, crown depth, height	Measurement						
		Foliar N, C, S	Dry Combustion Analyzer						
Foliar total Ca, Mg, K, Mn, Fe, Al, Cu, B, Zn	ICP-OES								
Inorganic S	HI-reduction; Ion Chromatography								
Species % cover and richness	Visual inspection								
Total elements (45)	ICP-MS								
Abundance, richness, vigour, and health	Visual inspection								
Vegetation (jack pines)		Every ~6 years; since 1998; grab sample							
Vegetation (understorey)									
Vegetation (lichens)									

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
A-MD-2 Deposition and Effects (Dabek, O'Brien)	Air	PM _{2.5} speciation: ions (e.g., NH ₄ ⁺ , K ⁺ , Ca ²⁺ , Mg ²⁺ , Na ⁺ , Cl ⁻ , NO ₃ ⁻ , SO ₄ ²⁻ , organic acids), OC/EC, biomass burning markers (e.g., levoglucosan), water-soluble elements Precursor gases (NH ₃ , HNO ₃ , HONO, SO ₂) Gases (NO, NO ₂ , HONO, NO _y , NH ₃ , SO ₂)	Sequential Met One Super SASS (sampling train)	24 hr, 1 in 3 days	AMS 1 (2015), AMS 4 (2017), AMS 18 (2015)	Evaluation the contribution of oil sands sources relative to other sources of total N and S deposition; evaluate CTMs; validate satellite observations		Continue monitoring until at least 2020 and perform data evaluation and for, e.g., deposition modeling, health impacts, apportionment
		PM _{2.5} speciation (K ⁺ , Ca ²⁺ , Mg ²⁺ , Na ⁺ , NO ₃ ⁻ , SO ₄ ²⁻ , Cl ⁻) Gases (SO ₂ , HNO ₃)	Continuous analyzers 3-stage filter pack	5-min average; since 2015; active Daily average; collected daily since 2015; active	Pinehouse Lake, SK (2015); Flat Valley, SK (2016)			
	Wet Deposition	Ions (Cl ⁻ , NO ₃ ⁻ , SO ₄ ²⁻ , Na ⁺ , NH ₄ ⁺ , K ⁺ , Mg ²⁺ , Ca ²⁺) and pH	Wet-only precipitation collector	Daily average; collected daily; since 2015				

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring	
WL-MD-10 Wetland Monitoring (Cobbaert)	Bulk Deposition	NH ₄ ⁺ , Mg ²⁺ , Ca ²⁺ , Na ⁺ , NO ₃ ⁻ , SO ₄ ²⁻	Ion Exchange Resins	Biannual average; since 2009	6 bog sites up to 130 km away (4 sites since 2009)	Publications Wieders et al., (2010, 2016a,b) Vile et al., (2013) Reports Wieders et al., (2012)	Explicit linkages with A-LTM-3; previously combined IER data for Wieders et al., (2016a) and possible co-location study to compare Wieders and WBEA IERs	Major component of project is to evaluate and integrate OS wetland projects; continue monitoring these parameters for at least 3 years with intent to increase spatial coverage	
	Air	Meteorology (precip, T, RH, hv, PAR)	WatchDog® Weather Stations	Hourly average; since 2009					
	Moss	Growth Total C, N, S	Cranked wire Combustion Analyzer	Jun/Jul/Aug; since 2009; grab sample	Developing OS LTM wetland network by integration: (i) WL-MD-2 (ii) WL-MD-4 (iii) WL-LM-5 (iv) WL-MD-11 (v) WL-IC-3 (vi) B-LTM-1	See Table 4.1 for more information since focus is on PACs and elements	Continue to monitor until at least 2020		
	Bog Porewater	pH, conductivity NH ₄ ⁺	pH probe Autoanalyzer	3-5 samples per growing season; since 2017					
	Shrubs	DIN, DON, DIC, DOC	TC/TN Analyzer	Annually; since 2017					
		NO ₃ ⁻ , SO ₄ ²⁻	IC	Annually; since 2017					
	Lichen	Growth Community structure Pigmentation	Reflectance Measurement	Annually; since 2017	21 wetland sites	Makar et al. (2018) Whaley et al. (2018) (submitted) Maps of simulated emissions, deposition, and air concentrations for all species and critical load exceedances	Explicit linkages with other components of A-MD-4 (Li and Aklilu)	Continue evaluating model against observations	
	WL-MD-11 Amphibian and Wetland Health (Pauli)	Wetland porewater	"Basic water quality information (e.g. T, pH, N, etc...)"	GEM-MACH	since 2011; grab sample	Entire AOSR			
		Modelling	Deposition and air concentrations of: SO ₂ , pSO ₄ ²⁻ , NO, NO ₂ , pNO ₃ ⁻ , NH ₃ , pNH ₄ ⁺ , Na ⁺ , K ⁺ , Ca ²⁺ , Mg ²⁺ , Cl ⁻		N/A				
	Deposition and air concentrations of: SO ₂ , pSO ₄ ²⁻ , NO, NO ₂ , pNO ₃ ⁻ , NH ₃ , pNH ₄ ⁺ , Na ⁺ , K ⁺ , Ca ²⁺ , Mg ²⁺ , HCl, Cl ⁻ , NH ₃ bi-directional exchange								

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
A-MD-4 Aircraft N/S Deposition (Li)	Air	SO ₂	Thermo 43i	Aug 2013	Downwind of OS facilities	Maps of observed SO _x deposition downwind of Oil Sands	Explicit linkages with other components of A-MD-4 (Makar)	Follow up Aircraft study in Apr/July 2018
		pSO ₄ ²⁻ , pNO ₃ ⁻	Aerodyne AMS					
		NO _x , NO _y	Thermo 42i					
		Total S deposition	TERRA + aircraft					
Critical Load of Acidity (Akliu)	Modelling	Critical Load Exceedances	Steady-state mass balance	N/A	Alberta	Province wide map of critical load values and exceedances	None	Additional sensitivity tests and comparison with CMAQ deposition

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Table B.2 Mercury and Trace Elements

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
A-MD-9-1718 Atmospheric deposition to lakes and snowpack	Snow, sediment cores	Hg, methyl Hg (MeHg), trace elements (45 elements)		2017/18: Collect paired snowpack samples in the open and under the forest canopy at 40 sites; collect sediment cores at 5 lakes in the Athabasca OS (same as PACs)	Athabasca OS, Peace Athabasca Delta (125 km of major OS developments)	Estimate Hg deposition and produce deposition maps Share data with industry partners (COSIA) to develop mitigation plans Study on spatial temporal patterns in trace element deposition from lake sediment cores Study sources of MeHg to snowpacks, methylation and particulates in snow Kirk et al., (2014): Map winter Hg, MeHg deposition as a function of distance from sources Validate Hg deposition model using snowpack data	Deposition data provided to AQRD (A. Dastoor, T. Harner, L. Zhang) to improve deposition modeling Study spatial temporal patterns in trace element deposition from lake sediment cores Kirk et al., (2014): Map winter Hg, MeHg deposition as a function of distance from sources Validate Hg deposition model using snowpack data	2017/18: Collect paired snowpack samples in the open and under the forest canopy at 40 sites; collect sediment cores at 5 lakes in the Athabasca OS (same as PACs)
A-MD-8-1718 Acid lakes monitoring	Lakes	Elements and Hg/MeHg		Sampled once annually each fall for the past 15 years (except Hg and MeHg since 2012)	50 small shallow lakes	Study regional drivers (ecological, climatological, hydrological, geological) of lake water chemistry and their spatial and temporal patterns		Sample collection each Fall

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
A-MD-2-1718 Deposition and effects	Air	Elements; PM _{2.5} , PM10-2.5	Elements: FRM instruments Dichot samplers	24 hr PM _{2.5} : 1 in 6 days (AMS 5, 11, 13, 17) PM ₁₀ : PM10-2.5: 1 in 6 days (AMS 17) PM ₁₀ : PM10: 1 in 3 days (AMS 1, 4, 18)	3 pilot sites (AMS 5, 11, 13) since Dec 2010, but terminated in 2015 (AMS 13) and 2016 (AMS 5) AMS 1, 17, 18 added in 2015 AMS 4 added in 2017	Phillips-Smith et al., (2017): source apportionment of elements Study PM composition over space and time Study petcoke dust impacts on metal concentrations (especially vanadium) Dry deposition modeling of trace elements	Dry deposition modeling of trace elements	5 new integrated monitoring sites starting in 2017 (AMS1, 4, 11, 17, 18)
	Air	Total gaseous Hg (TGM), speciated Hg (GEM, GOM, PBM)	Tekran instruments: Model 2537 for TGM, Models 2537/1130/1135 for speciated Hg	5 min TGM, 2-4 h speciated Hg	AMS6 (upwind), AMS13 (downwind)	Parsons et al., (2013): analysis of TGM variations Study Hg cycling by analyzing data from the two sites Study land use effect on Hg cycling Study uptake of Hg by forests	Provide air and wet dep data to water and wildlife groups for model development	TGM measurements at AMS6 Co-located TGM, speciated Hg and Hg wet deposition at AMS13 Wet deposition collected under forest canopy and in the open (similar to snowpack collection in A-FS-1-1718)
	Wet deposition	Total Hg in precipitation	Wet deposition collector		Co-located with air Under the forest canopy and in the open			

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
A-LTM-3-1718 Atmospheric pollutant deposition monitoring network – to forest	Air	PM _{2.5} composition (U, Sm, Be, Pr, Nb, Ti, Th, Cs, Bi, W, La, Ag, Nd, Sb, Ce, Co, Cd, Sn, As, Se, Sr, Mo, Cr, Rb, Pb, Ni, Ca, Li, Cu, Mn, V, Ti, Zn, Mg, Na, Ca, Fe, Al, K, Si)	Active sampling using annular denuder system	Monthly average	2011-present: 4 sites (JP104, JP107, JP204, JP213)		Provide ambient air data to estimate deposition to forests	2011-present: monitoring at 4 sites
B-LTM-3-1718 Ecosystem health monitoring – forest health								
WL-MD-11-1718 Amphibians and wetland health	Wetlands	Hg, As, Cd, Pb, Se	Water and lichens sampling	Since 2011	21 sites	Compare elements in water and tissues to CCME guidelines Study spatial patterns in elements in wetlands Study elements in different life stages of wood frogs		Intensive monitoring: elements in wetlands and wood frog (same as PAH, naphthenic acids; continuation of sampling since 2011) Extensive monitoring: integration with other OS monitoring programs and research
	Wood frogs		Tissue sampling					

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
B-MID-8-1718 Monitoring Hg in colonial waterbirds	Colonial waterbirds	Hg	Egg sampling	2009-2016	12 sites	Study spatial and temporal patterns in egg and fish Hg levels	Study relationship between atmospheric deposition and spatial patterns in egg Hg levels	Continue sampling and analysis of Hg in eggs and fish
	Fish	Hg	Tissue sampling	Since 2013	9 sites: 4 along Athabasca River, 5 from PAD, Lake Athabasca and Peace River PAD	Study factors driving Hg bioavailability (distance to sources, food web effects on biomagnification, atmospheric deposition)		MeHg monitoring in water and zooplankton, benthic invertebrates
	Water, zooplankton, benthic invertebrates	MeHg						
B-IC-WCT-1718 Monitoring contaminants and toxicity in wildlife	Waterfowl and aquatic furbearers (muskrat, mink, river otter)	Hg, MeHg, Sb, As, Cd, Cr, Cu, Pb, Ni	Collection of carcasses partnering with indigenous communities and commercial hunters/trappers and obtain tissue samples from wildlife Latrine site surveys: swab and collect fresh feces; store samples	Latrine survey: Mark old feces Visit site every 24 h	Athabasca OS surface mining, Peace Athabasca Delta, reference sites	Study the extent heavy elements bioaccumulate and biomagnify in food chain of boreal ecosystem and wildlife health impacts Population-level impact assessment study from DNA Study contribution of heavy elements in biota from oil sands sources, forest fires, and natural bitumen deposits	2017-20 (same as PAHs): collect waterfowl, aquatic furbearers (muskrat, mink, river otter), fish, and aquatic invertebrates for analysis of contaminants integrate wildlife collection with gulls, amphibians, wetland sites from other OS projects latrine site surveys for river otters	

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
A-MD-4 Mercury deposition modelling	Air and deposition	Hg; GEM, GOM, PBM	GEM-MACH-Hg model		Athabasca OS and across Canada	Modeling concentrations and deposition and contributions from various sources and provinces Modeling impact of biomass burning on concentrations and deposition Modeling deposition in the Athabasca OS and comparing with snowpack deposition	Modeling deposition and its contributions from various sources and provinces Modeling impact of biomass burning on deposition Modeling deposition in Athabasca OS and comparing with snowpack deposition (in A-FS-1-1718)	2017-20 (same as PAHs): collect waterfowl, aquatic furbearers (muskrat, mink, river otter), fish, and aquatic invertebrates for analysis of contaminants integrate wildlife collection with gulls, amphibians, wetland sites from other OS projects latrine site surveys for river otters

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Table B.3 Polycyclic Aromatic Compounds (PAC)

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
A-MD-9-1718 Atmospheric deposition to lakes and snowpack	Snow, sediment cores	52 routine analytes and novel PACs (alkyl PAHs, thia-arenes, aza-arenes)	Analytical: GC-MS and novel PACs analyzed by GCxGC-TOFMS (2D gas chromatography time of flight mass spectrometry)	2017/18: Collect paired snowpack samples in the open and under the forest canopy at 40 sites; collect sediment cores at 5 lakes in the Athabasca OS	Athabasca OS, Peace Athabasca Delta (125 km of major OS developments)	Estimate PAC deposition and produce deposition maps Share data with industry partners (COSIA) to develop mitigation plans Study of novel PACs in petcoke, road haul dust, bitumen	Deposition data provided to AQRD (A. Dastoor, T. Harner, L. Zhang) to improve deposition modeling	2017/18: Collect paired snowpack samples in the open and under the forest canopy at 40 sites; collect sediment cores at 5 lakes in the Athabasca OS
				2011/12: snowpack sampling along transects 2014/15: gridded snowpack sampling 2012-16: snow sampling at 10-12 sites near Fort Chipewyan 1910-2015: data from 25 lake sediment cores decadal averaged		Manzano et al., (2016a,b) Study on spatial and temporal patterns in deposition from dated sediment cores Study of spatial and temporal trends of novel PACs in snow, sediment and air PAC deposition maps during winter Study comparing novel PACs in snow, lake sediments and air with petcoke samples		

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
A-MD-2-1718 Deposition and effects	Air	PAHs, alkylated PAHs, DBTs, transformation products (oxy- & nitro-PAHs)	Passive air samplers	2-month integrated concentrations	Since 2011, 15-18 sites in Athabasca OS	Harner et al., (2013), Schuster et al., (2015), Hsu et al., (2015), Jariyasopit et al., (2015) Study evaluating PAC emissions inventory using dispersion modeling Map PAC deposition Study comparing transformation products to parent PACs and chamber studies Study in-vitro toxicity of air samples Study factors affecting PACs in air	Map PAC deposition Data provided to WL-MD-11	Continue to collect samples at all 18 sites but samples analyzed and reported at 5 sites (AMS5, 6, 9, 13, 14) starting in 2016/17
		PAHs, alkylated PAHs, DBTs	Active air samplers – high volume PUF and filters	24 hr; 1 in 6 days	3 pilot sites (AMS5, 11, 13) since Dec 2010; terminated in 2015 (AMS13) and 2016 (AMS5) AMS1, 17, 18 added in 2015 AMS4 added in 2017	Zhang et al., (2015a, b) Source apportionment study of coke dust impacts on PACs, source contribution of petcoke to BaP Study characterizing PAC and trends Study PAH transformation to quinones	Zhang et al., (2015a): analysis of wet scavenging efficiencies of PACs in rain and snow Zhang et al., (2015b): dry deposition modeling of PACs to various land covers	Previous 2 pilot sites terminated in 2015 (AMS13) and in 2016 (AMS5) 5 current integrated monitoring sites starting in 2011 (3rd pilot site AMS 11), 2015 (AMS1), 2016 (AMS17, 18), and 2017 (AMS4)
		VOCs, polar and sulfur-containing VOCs	Canisters and cartridges	24 hr; 1 in 6 days	3 pilot sites (AMS5, 11, 13) since Dec 2010; AMS13 (terminated in 2015) AMS05 (terminated in 2016) AMS1, 17, 18 added in 2015 AMS4 added in 2017	Study predominant emission sources, transformation		5 current integrated monitoring sites (AMS1, 4, 11, 17, 18)

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
A-MD-2-1718 Deposition and effects - cont'd	Air/Dry deposition	PAHs, alkylated PAHs, DBTs, transformation products (oxy- & nitro-PAHs)	Passive dry deposition samplers (PAS-DD) and PUF disk		Sampled for 1 year (Oct 2015-Sep 2016) at AMS5, 6, 9, 13, 14 in Athabasca OS	Study comparing PUF disk and PAS-DD samplers Study comparing PAS-DD sampling results to deposition models Study identifying petcoke dust signature from air samples	Study comparing PAS-DD sampling results to deposition models	Continue sampling in 2017-18 at 5 sites (AMS5, 6, 9, 13, 14), maybe extended to 2019
	Tree cores, bark and wood	PAHs, alkylated PAHs, DBTs, transformation products (oxy- & nitro-PAHs)			18 sites (same as passive) sampled in 2015	-Study of historical trends in PACs including before oil sands mining Source identification study of PACs from tree bark and wood		Continue tree sampling in 2017-19

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
WL-MD-11-1718 Amphibians and wetland health	Wetland	PACs	Water sampling	Multiple time points during 2011-17	21 sites since 2011	Compare PAHs in water to CCME guidelines Study dominant PAC species in wetland Study uptake of low and high MW PACs	Uses deposition data from A-MD-2	Continue sampling in 2017-18 at 5 sites (AMS5, 6, 9, -Intensive monitoring: PAHs, naphthenic acids in wetlands and wood frog (continuation of sampling since 2011) 13, 14), maybe extended to 2019 Extensive monitoring: integration with other OS monitoring programs and research
	Semi-permeable membrane devices (SPMD)	PACs		Deployed for 30 days during spring/summer 2013-17	13 sites since 2011	Study PACs in SPMD as a function of distance from upgraders Study dominant PAC species in SPMD Study in-vitro toxicity of SPMD extracts		
	Wetland sediment	PACs	Sediment core sampling		13 sites since 2011			
	Wood frog tadpoles, recent metamorphs (RM), adults	PACs			21 sites since 2011	Study PAC concentrations in tadpoles Study comparing PAC concentrations between tadpoles, RMs and adult wood frog Study of spatial variability of PAHs in frogs		

Project	Media Monitored	Parameters Monitored	Methodology	Sampling Frequency and Data Coverage	Sites	Publications/ Products that use these monitoring data	Existing Linkages to Other OSM Deposition Projects	Planned Monitoring
B-MD-8-1718 Monitoring Hg in colonial waterbirds (previous work on PACs)	Gull eggs	PAHs, alkylated PAHs, DBTs			Calgary, WBNP, Lake Athabasca, Peace River	Previous studies: Compare PAH, alkylated PAH, DBT in gull eggs Spatial alkylated PAH composition Relationship between PAH and mercury		No further plans to monitor PACs in gulls
B-MD-9-1718 Monitoring contaminants and toxicity in wildlife	Waterfowl and aquatic furbearers (muskrat, mink, river otter)	PAH, alkylated PAH	Collection of carcasses partnering with indigenous communities and commercial hunters/trappers and obtain tissue samples from wildlife Latrine site surveys: swab and collect fresh feces; store samples	Latrine survey: -Mark old feces -Visit site every 24 h	Athabasca OS surface mining, Peace Athabasca Delta, reference sites	Study exposure to alkylated PAH and health impacts (endocrine disruption) in river otters Population-level impact assessment study from DNA Study contribution of PAHs in biota from oil sands sources, forest fires, and natural bitumen deposits		2017-20: collect waterfowl, aquatic furbearers (muskrat, mink, river otter), fish, and aquatic invertebrates for analysis of contaminants integrate wildlife collection with gulls, amphibians, wetland sites from other OS projects latrine site surveys for river otters

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Appendix C – Acronyms and Abbreviations

AEP – Alberta Environment and Parks

alk-PAHs – alkylated polycyclic aromatic hydrocarbons

AMS – Air Monitoring Station

AOSR – Athabasca Oil Sands Region

Ca₂⁺ – calcium (either in particulate matter, precipitation, or surface reservoirs)

CALMET – diagnostic meteorological model that reconstructs 3D wind and temperature fields

CALPUFF – dispersion model used to simulate air pollutant transport

CAPMoN – Canadian Air and Precipitation Monitoring Network

CEMS – continuous emissions monitoring systems

CMAQ – Community Multiscale Air Quality model (a 3-D chemical transport model that can simulate air pollution concentrations and deposition)

CoTAG – conditional time averaged gradient

DBTs – dibenzothiophenes (a subset of PACs)

DIC – dissolved inorganic carbon

DIN – dissolved inorganic nitrogen

DOC – dissolved organic carbon

DON – dissolved organic nitrogen

ECCC – Environment and Climate Change Canada

EMSD – Environmental Monitoring and Science Division (a division within AEP)

FIA – flow injection analyzer

FS – focused study, designed to answer specific question(s) within a limited (<5 years) timeframe

GEM-MACH – Global Environmental Multi-scale Modelling Air quality and Chemistry (a 3-D chemical transport model that can simulate air pollution concentrations and deposition)

Hg – mercury (refers to all mercury-containing species)

Hg⁰ – elemental mercury

Hg^{II} – oxidized mercury

HI – hydrogen iodide

HNCO – isocyanic acid gas

HNO₃ – nitric acid gas
HONO – nitrous acid gas
hv – solar insolation (sunlight)
IC – ion chromatography
ICP-OES – inductively coupled plasma optical emission spectrometry
IER – ion exchange resin
K⁺ – (either in particulate matter, precipitation, or surface reservoirs)
LMWOA – low molecular weight organic acids
LTM – long-term monitoring
MeHg - methylmercury
Mg₂⁺ – magnesium (either in particulate matter, precipitation, or surface reservoirs)
MOUDI – micro-orifice uniform deposit impactor
N – nitrogen (refers to all nitrogen-containing chemical species)
N₂O₅ – dinitrogen pentoxide gas
NH₃ – ammonia gas
NH₄⁺(aq) – aqueous ammonium (either in precipitation or surface reservoirs)
NH_x – sum of NH₃ and NH₄⁺ in a system
NO₃⁻(aq) – aqueous nitrate (either in precipitation or surface reservoirs)
NO – nitric oxide gas
NO₂ – nitrogen dioxide gas
NO_x – gaseous nitrogen oxides (=NO + NO₂)
NOy – atmospheric oxidized nitrogen (refers to all forms of oxidized nitrogen in the atmosphere)
NPP – net primary productivity
OS – Oil Sands
OSM – Oil Sands Monitoring
P – phosphorous (refers to all phosphorous-containing species)
PACs – polycyclic aromatic compounds
PAHs – polycyclic aromatic hydrocarbons (a subset of PACs)

PAI – potential acid input

PAN – peroxyacetyl nitrate

PAR – photosynthetically active radiation

PLFA – phospholipid-derived fatty acids

PI – principal investigator

PM_{2.5} – fine particulate matter (smaller than 2.5µm in diameter)

pNH₄⁺ – ammonium in particulate matter

pNO₃⁻ – nitrate in particulate matter

PO₄³⁻ – phosphate (either in particulate matter, precipitation, or surface reservoirs)

PPN – peroxypropionyl nitrate

pSO₄²⁻ – sulphate in particulate matter

RH – relative humidity

S – sulphur (refers to all sulphur-containing chemical species)

SASS – speciation air sampler system

SO₂ – sulphur dioxide gas

SO₄²⁻(aq) – aqueous sulphate (either in precipitation or surface reservoirs)

SO_x – sulphur oxides (= SO₂ + HSO₄⁻ + SO₄²⁻)

T – temperature

TC –total carbon (refers to the sum of all carbon-containing species in a system)

TDN – total dissolved nitrogen (refers to the sum of all dissolved nitrogen species in a system)

TN – total nitrogen (refers to the sum of all nitrogen-containing species in a system)

TOC – total organic carbon

TS – total sulphur (refers to the sum of all sulphur-containing species in a system)

WBEA – Wood Buffalo Environmental Association

Appendix D – Workshop Agenda and Participant List

OSM Deposition Integration Workshop

AGENDA

Conference Room 3 (1S625), ECCC Downsview (4905 Dufferin Street, Toronto, ON)

DAY 1 – SEPTEMBER 27, 2017		
1:00 – 1:15	Welcome and Introductions	Greg Wentworth and Stewart Cober
1:15 – 2:00	Deposition Integration: Overview and Goals	Greg Wentworth
2:00 – 2:30	Roundtable Discussion: Framework Format	All participants
2:30 – 2:45	Health Break (refreshments provided)	
2:45 – 4:15	Break-out Discussion: Does the Framework appropriately summarize the current level of knowledge? Group A: Acidifying and Eutrophying Pollutants Group B: Metals, Hg and PACs	All participants
4:15 – 4:30	Day 1 Wrap Up	All participants

DAY 2 – SEPTEMBER 28, 2017		
9:00-9:10	Welcome and Announcements	Greg Wentworth and Leiming Zhang
9:10-10:15	Break-out Discussion (Report Back): What changes are required to the Framework? What recommendations could be made? Group A: Acidifying and Eutrophying Pollutants Group B: Metals, Hg and PACs	All participants
10:15-10:30	Health Break (refreshments provided)	
10:30-12:00	Break-out Discussion: What are the current and future data and integration needs? Group A: Deposition Monitoring and Modelling Group B: Receptor Monitoring	All participants
12:00-1:30	Lunch	
1:30-3:15	Break-out Discussion (Report Back): What are the current and future data and integration needs? Group A: Deposition Monitoring and Modelling Group B: Receptor Monitoring	All participants
3:15-3:30	Health Break (refreshments provided)	
3:30-4:15	Roundtable Discussion: Knowledge Gaps and Recommendations	All participants
4:15-4:30	Workshop Wrap Up	Greg Wentworth and Leiming Zhang

OSM Deposition Integration Workshop

Participant List

1. Yayne Aklilu EMSD, Government of Alberta
2. Danielle Cobbaert EMSD, Government of Alberta
3. Colin Cooke EMSD, Government of Alberta
4. Paul Drevnick EMSD, Government of Alberta
5. Thompson Nunifu EMSD, Government of Alberta
6. Greg Wentworth EMSD, Government of Alberta
7. Carla Davidson Endeavour Scientific, participating on behalf of WBEA
8. Eric Edgerton ARA Inc., participating on behalf of WBEA
9. Ken Foster Owl Moon Consulting, participating on behalf of WBEA
10. Matt Landis Integrated Atmospheric Solutions, participating on behalf of WBEA
11. Ellen MacDonald University of Alberta, participating on behalf of WBEA
12. Irene Cheng Atmospheric Science and Technology, ECCC
13. Stewart Cober Atmospheric Science and Technology, ECCC
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23. Patricia Chambers Water Science and Technology, ECCC
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26. Johan Wiklund Water Science and Technology, ECCC
27. Bruce Pauli Wildlife and Landscape Science, ECCC
28. Philippe Thomas Wildlife and Landscape Science, ECCC
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30. Kel Wieder Villanova University

