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Forest health effects due to atmospheric deposition: Findings from long-term forest health monitoring in the Athabasca Oil Sands Region

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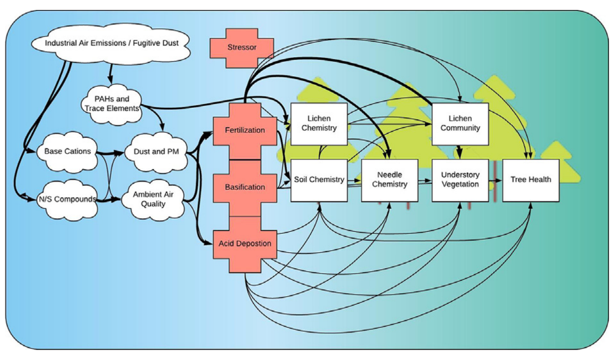
HIGHLIGHTS

- 20-year oil sands monitoring program designed to detect acidification in forests
- S, N and base cation deposition patterns reflect oil sands development.
- Dust is neutralizing acidification effect and deposits trace element and PACs.
- Evidence for fertilization, rather than acid deposition effect on forests
- Monitoring program conceptual model can be adapted to include fertilization

GRAPHICAL ABSTRACT



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ABSTRACT

Oil sands developments release acidifying compounds (SO_2 and NO_2) with the potential for acidifying deposition and impacts to forest health. This article integrates the findings presented in the Oil Sands Forest Health Special Issue, which reports on the results of 20 years of forest health monitoring, and addresses the key questions asked by WBEA's Forest Health Monitoring (FHM) Program: 1) is there evidence of deposition affecting the environment?, 2) have there been changes in deposition or effects over time?, 3) do acid deposition levels require management intervention?, 4) what are major sources of deposited substances? and 5) how can the program be improved?

Deposition of sulphur, nitrogen, base cations (BC), polycyclic aromatic compounds and trace elements decline exponentially with distance from sources. There is little evidence for acidification effects on forest soils or on understory plant communities or tree growth, but there is evidence of nitrogen accumulation in jack pine needles and fertilization effects on understory plant communities. Sulphur, BC and trace metal concentrations in lichens increased between 2008 and 2014. Source apportionment studies suggest fugitive dust in proximity to mining is a primary source of BC, trace element and organic compound deposition, and BC deposition may be neutralizing acidifying deposition. Sulphur accumulation in soils and nitrogen effects on vegetation may indicate early stages of acidification. Deposition estimates for sites close to emissions sources exceed proposed regulatory trigger levels, suggesting a detailed assessment of acidification risk close to the emission sources is warranted. However, there is no evidence of widespread acidification as suggested by recent modeling studies, likely due to high BC deposition. FHM Program evolution should include continued integration with modeling approaches, ongoing collection and assessment of monitoring data and testing for change over time, and addition of monitoring sites to fill gaps in regional coverage.

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1. Introduction

The Athabasca Oil Sands Region (AOSR) has been the site of intense industrial development since the 1960s, leading to concerns regarding the environmental effects, including air emissions and atmospheric deposition to the surrounding forest and local communities. In the mid-1990s, the Wood Buffalo Environmental Association (WBEA) established the Terrestrial Ecological Effects Monitoring (TEEM) Committee, which designed and implemented the Forest Health Monitoring (FHM) Program to address these concerns (Foster et al., 2019). Deposition in FHM plots is measured seasonally, and forest health endpoints including soil and needle chemistry, understory community composition and jack pine basal annual increment (BAI) are measured in intensive campaigns every six years.

There has been a great deal of research on the impacts of oil sands development to the receiving environment (reviewed in part in Foster et al., 2019), but few reported results on the long-term effects of deposition on forest health. A 2015 synthesis report by WBEA (Clair and Percy, 2015) assessed the results of an intensive sampling campaign in 2010/2011 and found evidence of deposition impacts to forest health. The work described within the Virtual Special Issue (VSI) on Forest Health Monitoring builds upon this work.

Studies from the FHM Program and others have shown that while there is widespread acidifying deposition, the concomitant deposition of base cations (BC) in soils limits effects (Watmough et al., 2014). BC are found in fugitive dusts that are prevalent in the mineable oil sands area, and are also a key driver of deposition of trace elements and polycyclic aromatic carbons (PACs) (Landis et al., 2017; Landis et al., 2019a; Landis et al., 2019b; Manzano et al., 2017; Shotyk et al., 2016a, 2016b; Zhang et al., 2016). At the same time, predictive models based on emissions predict widespread effects from acidifying deposition (Makar et al., 2018). This has led to public controversy about whether there is, in fact, widespread acid deposition from oil sands, and if so, the extent of the impacts on the health of the receiving environment.

The FHM Program was designed specifically to detect effects from acid deposition; this Forest Health VSI synthesizes nearly 20 years of forest health and deposition data to assess the key questions of the FHM Program (Foster et al., 2019). The FHM Program is based on a clear conceptual model outlining stressors and responses (Foster et al., 2019, Fig. 1) that supports the development of clear triggers to either focus monitoring on areas of higher risk or to require management action (Lindenmayer and Likens, 2009; Likens and Lindenmayer, 2018). Nitrogen and acidifying deposition due to industrial development are global issues (Bobbink et al., 2010). Recent research into critical loads for nitrogen deposition has found that earlier research may have overestimated critical loads due to complicating factors such as long term deposition history and forest harvest that altered the N cycle (Pardo et al., 2015; Simkin et al., 2019). The assessment of deposition effects in the AOSR provides a unique opportunity to observe these effects in forests with no harvest history, low baseline background levels of deposition, and inputs from relatively well-defined sources. This is somewhat unique among long-term forest monitoring approaches: other large forest health monitoring programs, including the European International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) and the United States Department of Agriculture Forest Health Monitoring Program (USDA FHM) are designed to maximize their applicability over large and varied geographies that may have multiple stressors. The ICP Forests program, which operates across 40 countries in Europe, incorporates a diversity of forest types and stressors and monitors endpoints such as tree condition, forest soil, foliar nutrient content and biodiversity. Only 10% of ICP Forests sites concurrently monitor stressors and effects (Galluzzi et al. 2018). The USDA FHM program focuses on characterizing status, changes and trends in indicators of forest condition (Potter and Conkling, 2017), and places a greater emphasis on detecting forest health change, but investigation of cause can be triggered if effects of sufficient magnitude are detected (Potter and Conkling, 2017).

In contrast, the FHM Program integrates stressor and effect monitoring facilitated by a clearly defined conceptual model of the stressor-effect pathway focused on acid deposition (Foster et al., 2019). There is a tension in environmental monitoring between monitoring all possible stressors and monitoring for effects (Dube, 2003; Dube and Munkittrick, 2003). The FHM Program focused on effects both important to local communities and

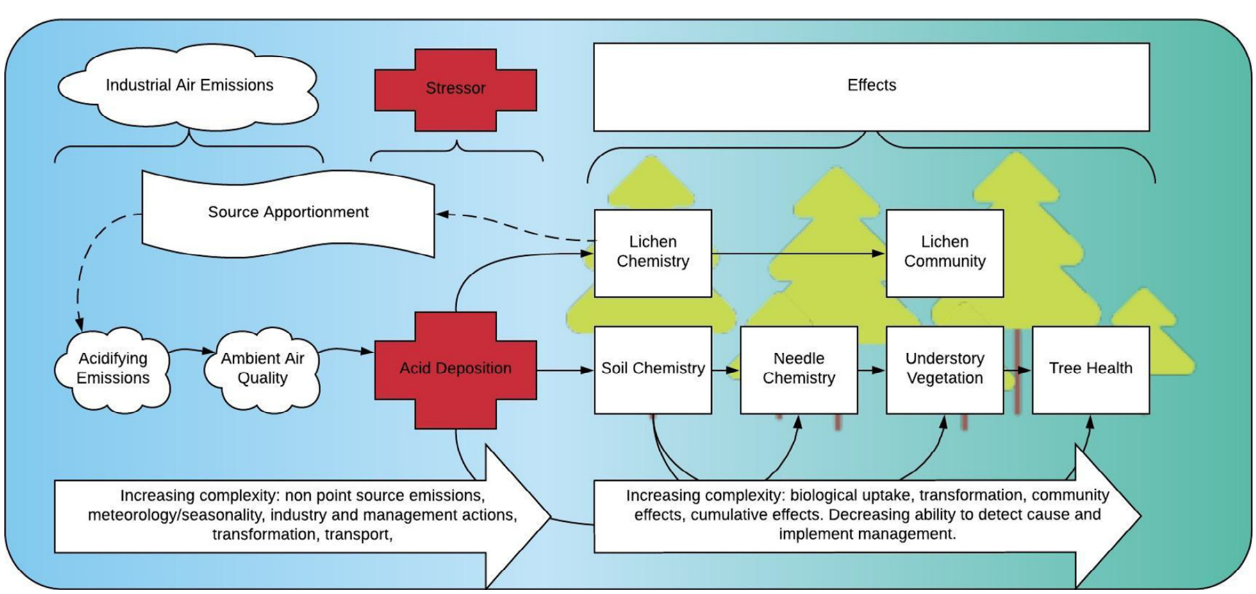

Fig. 1. Conceptual model for TEEM Forest Health Monitoring. The top line identifies the different conceptual parts of the monitoring program, from source (industrial air emissions) to stressor, to sink. Solid lines indicator cause-effect relationships, whereas the dotted lines show inferences determined from source apportionment studies (from Foster et al., 2019).

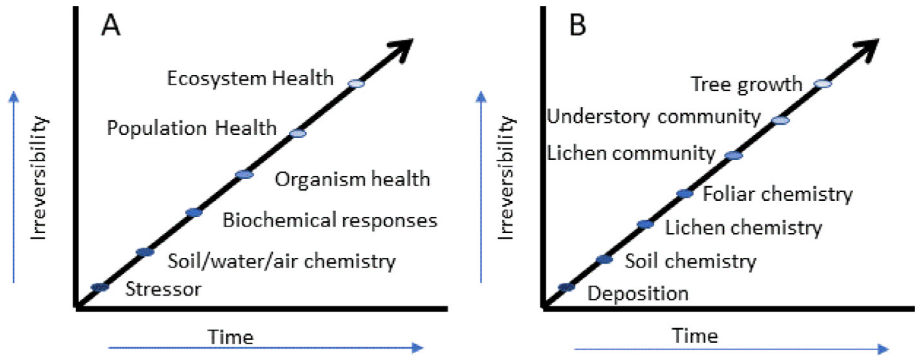

Fig. 2. Conceptual diagram showing levels of response that atmospheric deposition monitoring programs can address. Indicators further up the chain take longer to show effects and are harder to reverse. A: Conceptual diagram. B: Indicators for the TEEM Forest Health Program.

expected to be responsive to chemical deposition on reasonable time frames (Fig. 2A). Some soils in the AOSR are sensitive to acid deposition (Ok et al., 2007), therefore the program focuses on the acid-sensitive jack pine (*Pinus banksiana* Lamb.) forests on xeric, sandy soils (Foster et al., 2019). Over time, approaches to monitor deposition were added as they became available. The conceptual response model was that acidifying deposition would alter the soil chemistry (decrease pH, base cation:aluminum (BC:Al) and base saturation) and lichen chemistry. Once deposition exceeds a critical load, defined by the sensitivity of the receptor, the expected biological responses would include changes in lichen communities, understory communities, and/or tree growth (Fig. 2B) (Foster et al., 2019). The pairing of stressor and effects monitoring allows for continuous investigation of cause.

The work presented in the Forest Health VSI showed that there is deposition of acidifying compounds as well as BC, PACs and trace elements, and that these are correlated with changes in endpoints indicative of forest health. This paper synthesizes this work and builds on it to address five key questions: 1) is there evidence of deposition affecting the environment?, 2) have there been changes in deposition or effects over time?, 3) do acid deposition levels require management intervention?, 4) what are major sources of deposited substances? and 5) how can the program be improved?

2. Is there evidence of deposition affecting the environment?

2.1. Nitrogen and sulphur deposition

Developing a reliable estimate of the deposition of airborne compounds of concern, particularly gaseous and particulate N and S species and particulate matter, is a key objective of the Oil Sands Monitoring Program (Alberta and Canada, 2012). Edgerton et al. (2019) calculated gradients of deposition to jack pine forests throughout the AOSR using data from WBEA's ion exchange resins and passive monitoring network co-located at forest health sites. These analyses showed spatial and temporal variation of several parameters across the region. A clear influence of oil sands industrial operations is indicated by calculated deposition fields showing highest deposition of NH_4^+-N, NO_3^--N, and SO_4^{2-}-S coincident with mining activities and that levels decline to background within 50 km from the nearest source. Despite the difficulties and uncertainties associated with calculating deposition fields, the influence of the oil sands mines is apparent.

Total N deposition ranges from 2.0 to 5.7 kg ha^{-1} a^{-1}. Ambient NO_2 concentrations are high near emission sources and are a contributor to dry deposition in these areas. NH_4^+-N and NO_3^--N contribute to both wet and dry N deposition but have a much less pronounced gradient (Edgerton et al., 2019). Sulphur deposition ranges from 1.9 to 14 kg ha^{-1} a^{-1} and also declines to background levels within 50 km of the nearest emissions source (Edgerton et al., 2019). Its solubility is pH dependent and at the pH of rain observed in the oil sands region (6–7), SO_2 can be scavenged by rainwater and deposited as wet deposition (Jacob, 1999). SO_2 dry deposition depends greatly on surface characteristics. For needles that already have some amount of Ca^{2+}, likely close to mining and roads, this may be a relevant pathway for deposition, but at this time we do not have enough understanding of the dynamics of Ca^{2+} deposition. SO_4^{2-} is a dominant anion in fine particulate matter, and this may be a major contributor to S deposition (Edgerton et al., 2019).

Coincident with N and S deposition fields is deposition of Ca^{2+} and other BC via coarse and fine particulates, which may be neutralizing acidifying deposition (Watmough et al., 2014). Ca^{2+} deposition ranged from 2.21 to 11.28 kg ha^{-1} a^{-1} in a pattern similar to N and S deposition. Based on these data, Edgerton et al. (2019) predicted Potential Acid Input ($PAI = S_{deposition} + N_{deposition} - BC_{deposition}$, expressed in keq ha^{-1} year^{-1}) ranging from 0.1 to 0.2 keq ha^{-1} a^{-1} throughout the region, except for two clusters of sites in the heart of oil sands development (Edgerton et al., 2019). Dust deposition represents a significant component of deposition in the region (Landis et al., 2017; Fenn et al., 2015) and to date, has largely been an underappreciated dynamic affecting overall deposition and forest health in the oil sands region. Though this dust may be neutralizing acidifying deposition, it does not alter the fertilizing potential of N deposition and is itself a source of nutrients and other compounds of concern.

2.2. Trace elements and organic compounds

Concerns have also been raised about atmospheric deposition of material other than acidifying or fertilizing compounds, including trace elements (Bari et al., 2014; Blais and Donahue, 2015; Cumulative Effects Management Association, 2001; Golder Associates Ltd., 2003; Protano et al., 2014; Shotyk et al., 2017a, 2017b; Shotyk et al., 2014; Shotyk et al., 2015; Shotyk et al., 2016a, 2016b) and PACs (Bari et al., 2014; Cheng et al., 2018; Cho et al., 2014; Droppo et al., 2018; Fernie et al., 2018a; Fernie et al., 2018b; Hall et al., 2012; Harner et al., 2018; Jariyasopit et al., 2016; Kelly et al., 2009; Landis et al., 2015; Landis et al., 2016; Lundin et al., 2015; Manzano et al., 2016; Parajulee and Wania, 2014a, 2014b; Qiu et al., 2018; Schuster et al., 2015; Studabaker et al., 2017; Wnorowski, 2017; Zhang et al., 2016). Recent research has shown that fugitive dust is a vector for trace elements (Blais and Donahue, 2015; Shotyk et al., 2014; Shotyk et al., 2015; Shotyk et al., 2016a, 2016b) and PACs (Ahad et al., 2014; Landis et al., 2015; Landis et al., 2016; Parajulee and Wania, 2014a, 2014b; Schuster et al., 2015; Wnorowski, 2017). Zhang et al. (2016) found that stockpiles of petroleum coke represent a key source of PACs, with elevated deposition downwind of these stockpiles. PACs have also been detected in fine particulate matter (Landis et al., 2019a), lichens (Landis et al., 2019b), snowpack (Cho et al., 2014; Manzano et al., 2016) and in a variety of mine site $PM_{2.5}$ and PM_{10} dust sources (Wang et al., 2015).

In this issue, assessment of snowpack, lichen and $PM_{2.5}$ data all suggest that PACs are enriched in proximity to oil sands operations, and decline to background concentrations within 50 km of the nearest source (Landis et al., 2017; Landis et al., 2019a; McNaughton et al., 2019). This is a reduction of previous estimates of the range of the deposition fields (Manzano et al., 2016), due to clarification of source emissions, rather than using an arbitrary point in the middle of mining operations (Kelly et al., 2010; Kelly et al., 2009; Kirk et al., 2014). However, there is a temptation to consider distance to the nearest operation without considering the cumulative effects of multiple sources. Given the density of development, attributing a "zone of influence" to a particular source is not particularly meaningful; regional deposition estimates (Edgerton et al., 2019; Landis et al., 2019c) integrate multiple measurements and better highlight the scale of deposition regionally.

Stachiw et al. (2019) found that lingonberries, cranberries and blueberries close to oil sands had a higher concentration of trace elements such as Al, Cr, Pb, U and V deposited by atmospheric deposition, whereas Ba, Cd, Cu, Mn, Mo, Ni, Rb, Sr and Zn were obtained by uptake from the soil. Berries close to oil sands operations had higher concentrations of dust-borne elements compared to remote locations, and the trace element composition of this dust was consistent with a geological origin (soils and rocks). Deposition of these compounds was attributed to dust likely generated by industrial operations, which includes emissions from roads, exposed dry tailings, petroleum coke piles, but not from stacks, mine fleets or transportation (Stachiw et al., 2019). The authors found that rinsing berries removes most dust and associated metals. However, Fort McKay Community Based Monitoring shows that washing appears to decrease concentrations of some, but not all, dust-borne metals. It also appears that antioxidant activity may be altered by the act of washing berries but this depends on the metal and type of berry (Olsgard, personal communication, 2019).

While there is an emerging consensus that dust is an important factor in deposition dynamics in the AOSR, the amount of dust being deposited, its deposition pattern and its effect on acidifying input remain poorly understood (Edgerton et al., 2019; Makar et al., 2018), as are the deposition range and relative concentration of different trace elements (Manzano et al., 2016; Shotyk et al., 2016a, 2016b; Shotyk et al., 2017a, 2017b; Shotyk et al., 2014; Shotyk et al., 2015; Stachiw et al., 2019; Landis et al., 2017; Landis et al., 2019a; Landis et al., 2019b; McNaughton et al., 2019) and PACs (Landis et al., 2019b; Zhang et al., 2015; Zhang et al., 2016; Manzano et al., 2017; McNaughton et al., 2019) in dust. Some authors have also suggested that dust control could be a key mechanism to mitigate this deposition (McNaughton et al., 2019; Zhang et al., 2016), which, despite its difficulty, has recently become a recommended mining mitigation to reduce the effects of certain compounds in the receiving environment (CEAA, 2019). Reduced BC deposition through dust control could, however, have unintended consequences on acid deposition, because the loss of this buffering capacity without a concomitant reduction in acidifying deposition could tip the balance to increased acid deposition effects. Therefore, a better understanding of the quantities, characteristics and deposition patterns of regional dust emissions is imperative.

2.3. Ecological effects

MacKenzie and Dietrich (2019) and Bartels et al. (2019) analyzed soils and needles chemistry, understory plant communities and basal area increment of jack pine trees to assess the influence of total deposited N, S and Potential Acid Input (PAI) on forest health. Concentrations of N and S in jack pine needles were positively correlated with deposition, and soil concentrations of S were correlated with deposition, but not soil concentrations of N (MacKenzie and Dietrich, 2019). There was little evidence of PAI having any effect on soil pH (Cho et al., 2019; MacKenzie and Dietrich, 2019); however, determining changes in soil chemistry is very challenging and this warrants further exploration (Paragon Consulting, 2008).

Bartels et al. (2019) found conflicting evidence of relationships between nitrogen and sulphur deposition and jack pine growth rate. While jack pine growth rate is positively associated with being closer to oil sands development, this is likely the result of better growing conditions in the Athabasca River valley, an underlying pattern that existed prior to oil sands development. There is evidence of a general decline in growth rate since the beginning of the study period, but it is not clear whether this is associated with deposition or simply due to tree aging. There is, however, evidence that N and S deposition are associated with increased understory cover and richness, but little evidence of any effect of soil acidification (Bartels et al., 2019). Bartels et al. (2019) detected a potential effect of N and/or S deposition on understory richness, cover and diversity, but not an effect of BC or PAI. However, they were unable to distinguish the effects of S and N, as their deposition pattern covaries across the AOSR, though it's likely that it is the N deposition that alters species composition, as confirmed by experimental addition studies (Kwak et al., 2018). Earlier analysis of FHM Program data also showed a correlation between deposition of N, S and BC with increased cover and richness of vascular plants and a shift in species composition (Macdonald, 2015), indicating that fertilization may be occurring.

The effect of N deposition on forest growth is well established (Bobbink et al., 2010). Acidification and fertilization are also serious issues globally and widespread fertilization effects have been demonstrated in other forest monitoring programs (Bobbink et al., 2010). Bouwman et al. (2002) estimated that 7–17% of natural ecosystems globally have exceeded critical loads for acidification, and 7–18% of global natural ecosystems have exceeded critical loads for fertilization (Bouwman et al., 2002). In this VSI, Hemsley et al. (2019) estimated N deposition on reclaimed sites, which are much closer to emissions sources (within 500 m) than the TEEM forest health sites at 25 kg N ha^{-1}, dominated by NH_4^+-N. This is much higher than at the TEEM FHM Program sites, where a maximum deposition of 5.7 kg N ha^{-1} a^{-1} has been estimated (dominated by NO_2 and NH_4^+). Reclaimed jack pine sites were found to be at N saturation, because jack pine has a lower requirement for N and prefer to take up NH_4^+, which is quickly converted to NO_3^- in the soil (Hemsley et al., 2019). Similar to findings by MacKenzie and Dietrich (2019), Hemsley et al. (2019) found that excess N does not accumulate in the soils, but instead is absorbed by the roots and by the canopy and accumulates in the needles or is lost through leaching. Watmough et al. (2016) found similar results in a study of N-addition to jack pine forests but found that the major reservoir for N deposition was the lichen and moss understory. The lack of accumulation in the soil indicates that jack pine stands do not recycle N as effectively as aspen and may lose N through leaching (Hemsley et al., 2019). This work represents a useful upper bound of a likely critical load of 25 kg N ha^{-1} $year^{-1}$. A recent five-year experimental study was conducted to evaluate the response of jack pine forest to elevated levels of nitrogen (N) deposition (Watmough et al., 2016). N was applied as ammonium nitrate above the forest canopy by helicopter in an aqueous spray equivalent to a 0.06-mm rain event at dosages of 5, 10, 15, 20 and 25 kg N ha^{-1} $year^{-1}$ above background. Adverse ecological effects started at deposition rates of 10 kg N ha^{-1} $year^{-1}$. Similarly, application of 30 kg N over eleven years in a mixedwood boreal forest showed changes in herb diversity (Kwak et al., 2018). However, long-term (>10 years) studies are needed to fully evaluate N dynamics in forest ecosystems. Other research has shown that N deposition in nitrogen-limited forests like the boreal forest of 5–10 kg N ha^{-1} $year^{-1}$ affect soil mycorrhizae and plant communities, but again, data for forests with low N background deposition levels are lacking (Pardo et al., 2015; Simkin et al., 2019).

Similar work in ombrotrophic bogs, which receive all their nutrition from atmospheric deposition, also shows plant community responses that indicate fertilization in response to elevated deposition (Vitt et al., 2003; Wieder et al., 2011; Wieder et al., 2016a; Wieder et al., 2016b; Wieder et al., 2010). N deposition was shown to increase sphagnum primary production at low levels of deposition, but production decreased at 14.8–15.7 kg N ha^{-1} a^{-1}, indicating a critical threshold for N stress (Vitt et al., 2003). However, because low concentrations of N differentially stimulate growth of different species, subcritical loads of N can trigger community changes (Pardo et al., 2015). Deposition of N, S, Ca and Mg altered bog function, with accumulations of N and S in surface peat, and increased growth in *Sphagnum fuscum* (Wieder et al., 2016a). Five years of experimentally applied N (as NH_4NO_3) to a bog in the AOSR identified several ecological effects associated with N deposition (Wieder et al., 2019). To prevent the inhibition of N_2 fixation by increasing N deposition, Wieder et al. (2019) suggested that the critical load for N deposition in northern Alberta bogs should be 3 kg N ha^{-1} $year^{-1}$.

Critical loads for N deposition in jack pine soils are not known; however, estimates of critical loads in boreal forests for soil acidification may be as low as 1.62 kg N ha^{-1} $year^{-1}$ (Whitfield et al., 2011) and between 4 and 10 kg N ha^{-1} $year^{-1}$ for fertilization effects (Pardo et al., 2015; Simkin et al., 2019). The finding that understory plant communities respond at deposition levels at or below 5.7 kg N ha^{-1} $year^{-1}$, especially in an area with low background deposition, is within the range reported for other North American forests and suggests that further work to establish a management or monitoring trigger for N deposition is warranted (Cumulative Environmental Management Association, 2008).

3. Have there been changes in deposition or effects over time?

Earlier work has shown that higher percentile ambient concentrations of NO_2 are increasing (Bari and Kindzierski, 2015; Davidson and Spink, 2018) and that SO_2 has shown dynamic changes over the last 16 years of monitoring (Davidson and Spink, 2018). Furthermore, satellite data indicate that NO_2 levels in the region have been increasing and SO_2 levels have shown a slight downward trend (McLinden et al., 2016). Passive monitoring data collected between 2000 and 2017 also showed that the median concentration of ambient SO_2 has generally decreased, but shows little evidence for change in median NO_2 (Edgerton et al., 2019). However, linear trends analysis of measures of central tendency may not be the most sensitive method to detect change (Davidson and Spink, 2018); a more detailed analysis of higher and lower percentiles of the ambient concentrations measured by passive monitors may increase understanding of the dynamic nature of ambient air SO_2 and NO_2 concentrations, and deposition estimates based upon them.

Assessing change of deposition and ecological endpoints over time has proven challenging. In earlier sampling campaigns within the FHM Program, soil samples were composited to reduce costs. This sort of compositing is common in soil studies, and while it does not affect the ability to assess change in space within a given year, the loss of replication at each site reduces the ability to detect change over time at that location. Coupled with the loss of several sites to forest fires and industrial expansion (Foster et al., 2019), the FHM Program has had difficulty assessing change over time at forest health sites. As a result, several Forest Health VSI papers focused on analyses of data from the last intensive forest health campaign in 2011/2012. As additional data become available, the ability to assess ecological change over time related to changing deposition will improve, but will still be constrained by the lack of soil sample replication in early forest health campaigns. The creation of change management procedures and firm standard operating procedures that maintain sample integrity to answer questions of change over both time and space is imperative for the program. Changes in field sampling protocols should be rigorously examined and involve consulting by a statistician to ensure that the goals of the program are not compromised.

Despite this limitation, the FHM Program's several lichen collections and consistent analytical approaches have allowed the assessment of change over time over the study domain, showing increases in S, trace elements, and stability in N deposition between 2008 and 2014 (Landis et al., 2019c). Landis et al. (2019c) developed deposition fields using chemical analyses from lichen collections throughout the AOSR in 2002, 2004, 2008, 2011, 2014 and 2017. They similarly found that the influence of dust increased, particularly within 25 km of oil sands operations. Lichen S content showed the greatest increase between 2008 and 2014, the years of the most comprehensive collections, while N levels reflected the progression of industrial development, particularly in the north and east areas of the AOSR. The aerial extent of dust influence also increased, reflecting the expansion of oil sands operations between these years (Landis et al., 2019c). The increase in S detected by lichen collections contrasts with results from passive monitoring, which found generally decreasing ambient concentrations of SO_2 (Edgerton et al., 2019).

Given contrasting findings in trends in ambient concentrations, deposition and lichen concentrations, ongoing monitoring to support a detailed assessment of change over time to better understand the link between emissions, ambient air concentrations, deposition and effects is warranted.

4. Do acid deposition levels require management intervention?

The FHM Program was initially focused on acidifying deposition associated with oil sands development, thus calculating PAI to evaluate against defined critical loads is a primary goal of the program. The original management framework for acid deposition in Alberta was developed in 1999 (Foster et al., 2001), updated in 2008 (Government of Alberta, 2008), and is currently undergoing an update (Spink, personal communication, 2019). Grid cells, defined by 1° latitude × 1° longitude boundaries (locked such that whole degrees intersect at the centre of the grid cell), were assigned a critical load of 0.25, 0.50 or 1.00 keq ha^{-1} $year^{-1}$ based on their soil and surface water acidification sensitivity as interpreted from then-available province-wide datasets. Target loads, assigned to be 90% of the critical loads, were established as provincial regulatory objectives. Monitoring loads established at 70% of the critical loads were established as triggers for investigative actions designed to validate or adjust assigned critical loads and/or to more precisely estimate acidifying deposition. In 2004, a regional acid deposition management framework based on a dynamic deposition and response modeling approach was developed for the AOSR (Cumulative Environmental Management Association, 2004). Time-to-effects analyses requiring data-intensive dynamic modeling approaches, such as the Model of Acidification of Groundwater in Catchments (MAGIC) (Whitfield et al., 2010) can be used to predict acidification-induced effects on a 15 to 30 year timeframe. The MAGIC integrates data describing soil physicochemical properties with deposition estimates based on historic, current and predicted future emissions. A 2010 application of this model, using weathering rates that represent regional soils, 2010 deposition levels, complete N retention, and a very conservative acidification threshold (BC:Al = 10), indicated that 34% of forest sites were at risk of acidifying (Whitfield et al., 2010). A regional interim N (fertilization) management framework (Cumulative Environmental Management Association, 2008) was completed in 2008, but not adopted by regulatory agencies and did not set critical loads for N deposition, citing the need for further research completed in 2016 (Watmough et al., 2016).

Acid deposition assessments conducted during the development of the 2001 framework (using 1995 emissions inventories and RELAD modeling) indicated that none of the provincial grid cells were receiving deposition in excess of their assigned monitoring loads. Provincial-level deposition assessments in 2004 (WBK and Associates Inc., 2006) and 2011 (Alberta, 2014) continued to indicate that deposition within AOSR grid cells had not reached the assigned 0.17 keq ha^{-1} $year^{-1}$ monitoring load that would trigger focused examinations of receptor sensitivity. Though we have not done the RELAD modeling here, the point estimate of PAI (Edgerton et al., 2019) has exceeded the monitoring load for 19 of the 62 Forest Health sites in proximity to the emissions sources. Eight FHM sites, all near mining and upgrading operations, were identified as receiving >0.25 keq ha^{-1} $year^{-1}$, the critical load assigned to sensitive soils in the Alberta framework (Edgerton et al., 2019, Fig. 2). According to the current Acid Deposition Management Framework (2008), a PAI above 0.17 keq ha^{-1} $year^{-1}$ as determined by RELAD modeling over a grid cell measuring 0.5 deg latitude $\times$ 0.5 deg longitude requires more in-depth assessment of receptor (soil, water) sensitivities and predicted receptor responses to acidifying deposition.

No statistically significant decreases in BC:Al or base saturation were found in the analyses of the FHM soil samples collected in 2011–2012 (MacKenzie and Dietrich, 2019), however, there is high variability within these indicators. No acid deposition-related effects on jack pine tree growth or understory composition were found (Bartels et al., 2019). The absence of acid-induced effects in the soil and vegetation at acid-sensitive jack pine monitoring sites supports the conclusion that there are few to no indications of broad regional environmental acidification, which is consistent with other findings (Cathcart et al., 2016; Cho et al., 2019; Jung et al., 2013; Watmough et al., 2009). However, at FHM sites nearer to emission sources, sulphate content in mineral soils to depths of 30 cm was elevated (MacKenzie and Dietrich, 2019), possibly indicating early stages of soil acidification, which is consistent with studies that suggest sensitive soils across the region may be at risk of acidification over time (Cho et al., 2017b; Makar et al., 2018; Ok et al., 2007).

At the same time, NO_2 emissions are expected to lead to increased acid deposition (Cho et al., 2017a, 2017b) and recent analyses based on Environment and Climate Change Canada's GEM-MACH model suggest that a very large area of northeastern Alberta and northwestern Saskatchewan may be subject to exceedances of acid deposition critical loads (Makar et al., 2018). When applied to critical loads derived for mineral soils using BC:Al ratios of 2 (coniferous forest), 6 (mixed and broadleaf forest, shrubland) and 40 (grassland), Makar et al. (2018) suggested that deposition in excess of these receptor-specific critical loads may be occurring in and beyond the AOSR. There are significant differences between the GEM-MACH modeling approach and the analysis done by Edgerton et al. (2019), which used calculated deposition based on ion exchange resin measurements throughout the AOSR rather than predictive modeling. The two key areas of differences are likely an underestimation of BC deposition used in GEM-MACH analyses, and a much different approach to determining critical loads that provides a very conservative lens. Dust, the main vector of BC deposition, is very difficult to model from a bottom up approach given there are few point source emissions, many sources large in area, the components of dust vary by source and distance from source, and there are poor inventories for its emissions.

Another area of uncertainty, the critical load for acidification or sensitivity of the soils to acidification, also requires further work. Soil weathering rates are key determinants of the sensitivity of soils to acid deposition, and Abboud et al. (2002), Whitfield et al. (2010), and Whitfield et al. (2011) suggest that some regional soils may be very sensitive to acidifying deposition, due to very low weathering rates. The analysis of Makar et al. (2018) used NEG-ECP and CLRTAP protocols to assess critical loads, which may provide more conservative estimates of critical loads than established in the current Acid Deposition Management Framework (2008). Using a more conservative trigger of 0.1 keq ha^{-1} $year^{-1}$ to account for this potentially increased sensitivity and using point estimates of PAI, 56 of 62 FHM Program sites would have exceeded this level (Fig. 3). Focused studies could assess the impacts of deposition on these soils and others with weathering rates of <0.1 keq ha^{-1} a^{-1}, in keeping with the FHM Program principles of early warning and sensitive receptor-based effects monitoring. A modeling and rigorous monitoring program that complement each other are required to understand and manage acid deposition. More work is required to reconcile the contrasting predictions between the GEM-MACH model and the findings of the FHM Program.

5. What are major sources of deposited substances?

Five papers (Landis et al., 2019a; Landis et al., 2019b; Landis et al., 2019c; McNaughton et al., 2019; Stachiw et al., 2019) in this VSI described different approaches to investigate the sources and source types that are primary contributors to deposition in the AOSR. Despite the different approaches used, the identification of the major drivers of deposition of PACs, trace metals, and BC are consistently shown to be dust and particulate matter emissions (Fig. 4). The term "fugitive dust" has been used by different authors to include or exclude different sources; for the purposes of this paper the term includes dust from mine faces, road salt, biomass burning, petroleum coke piles, construction, haul roads, and wind-blown soils and tailings.

Between 2010 and 2011, daily integrated $PM_{2.5}$ and $PM_{10-2.5}$ samples were collected in Fort McKay, a First Nations and Métis community surrounded by oil sands mining operations, some within a few kilometres (Landis et al., 2017). Source apportionment found that 96% of $PM_{2.5}$ mass was explained by five factors: oil sands upgrading (32%), haul road dust (26%), biomass combustion (25%) long-range Asian transport lead source (9%) and winter road salt. Source apportionment also showed that 99% of the PM_{10} data was explained by six sources: haul road dust (40%), wind-blown oil sands (27%), a mixed source fugitive dust (16%), biomass combustion (12%), mobile source (3%) and a local copper factor (1%) (Landis et al., 2017). NH_3, and NO_x were found in $PM_{2.5}$ and PM_{10} fractions characterized as biomass combustion, while ambient NOx associates with the $PM_{2.5}$ fraction characterized as road salt. This is consistent with N production during forest fires and by mobile sources such as automobiles and mine fleets. S compounds were found in the oil combustion/bitumen upgrading source (as sulphur compounds in $PM_{2.5}$ and ambient SO_2), mixed dust and fugitive oil sand (as S compounds in PM_{10}). BC were primarily found in mixed dust sources, including haul road dust (Ca, K and Mg in PM_{10}), dust from dry exposed tailings and mine faces (Ca, K, and Mg in $PM_{2.5}$), biomass combustion (K in $PM_{2.5}$), road salt (Mg and Na in $PM_{2.5}$), fugitive haul road dust (Ca, K, and Mg in

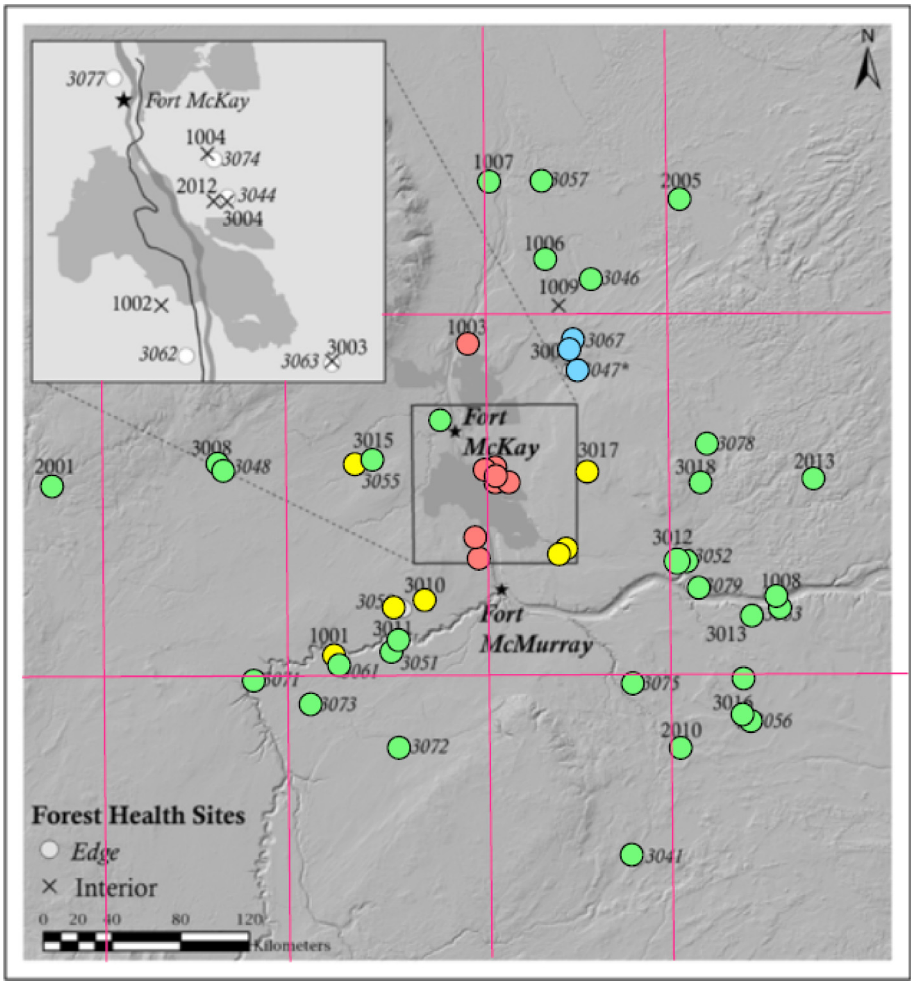

Fig. 3. Predicted Potential Acid Input (PAI) for Forest Health Sites as calculated by Edgerton et al. (2019). Pink = PAI > 0.25 keq ha^{-1} $year^{-1}$, yellow = 0.17 > PAI > 0.25 keq ha^{-1} $year^{-1}$, green = PAI < 0.17 keq ha^{-1} $year^{-1}$, blue = PAI < 0 keq ha^{-1} $year^{-1}$.

PM_{10}) and fugitive oil sands source (K, Mg, and Na in PM_{10}) (Landis et al., 2017). This is consistent with earlier source apportionment studies that showed windblown soil, biomass burning, haul road dust, soil and upgraders were sources of S and particulate matter that become significant sources of BC (Phillips-Smith et al., 2017).

This particulate matter is likely an important source of total S deposition (Edgerton et al., 2019), but less so of N deposition. The dominant measured ions in $PM_{2.5}$ are SO_4^{2-} and NH_4^+, and bulk deposition of NH_4^+-N and NO_3^--N accounts for almost half of the contribution to N deposition (Edgerton et al., 2019). Dry deposition of NO_2 dominates closer to the emissions source, but at more distal sites HNO_3, NO_2 and to a lesser extent NH_3 are equal contributors to dry deposition. Similarly, Hsu et al. (2016) showed that dry deposition of NO_2, and NH_3 were highest close to oil sands and urban emission sources, and Hemsley et al. (2019) found NH_3 to be the dominant component of deposition closest to the mines, while HNO_3 showed wider ranges of concentrations and a larger spatial extent around sources.

Particulate matter deposition in Fort McKay was also assessed for PAC loading and source apportionment, showing that major sources are biomass combustion (40%), fugitive dust (28%), upgrader stack emissions (21%), petrogenic PAH (18%) and transported aerosol (6%). Larger sized dust particles ($PM_{10-2.5}$) were explained by a five-factor PMF model with haul road dust (53%), mixed fugitive dust (32%), fugitive oil sand (10%), mobile sources (2%) and organic aerosol (1%) describing 98% of the data (Landis et al., 2019a). To assess PAH distribution over a larger area, Landis et al. (2019b) performed source apportionment using lichens distributed within a 150-km radius from the centre of oil sands development. They similarly found that petroleum coke and raw oil sands dust are the major sources of PACs, with biomass burning and mobile sources being secondary sources. Ninety percent of PACs were deposited within 25 km of the nearest industrial source (Landis et al., 2019b). Interestingly, only organic aerosols contributed to the total loading of PAHs in the $PM_{10-2.5}$ fraction, and pyrogenic PAH contributed to the total loading of PAHs in the $PM_{2.5}$ fraction (Landis et al., 2019b).

Several studies have assessed the prevalence and deposition of trace elements, including metals, to the receiving environment. This continues to be a controversial issue, however there is some emerging consensus that, again, dust is a major source of trace element deposition (Bari et al., 2014; Blais and Donahue, 2015; Cumulative Effects Management Association, 2001; Golder Associates Ltd., 2003; McNaughton et al., 2019; Shotyk et al., 2017a, 2017b; Shotyk et al., 2014; Shotyk et al., 2015; Shotyk et al., 2016a, 2016b; Vingiani et al., 2015; Zhu and Gueguen, 2016) (Fig. 4). One paper in this VSI assessed sources of Pb in particulate matter measured at Fort McKay and deposited to lichens across the AOSR and indicated that Pb deposition was attributed to eastern Asia (34%), local AOSR (20%) western Canadian (19%)

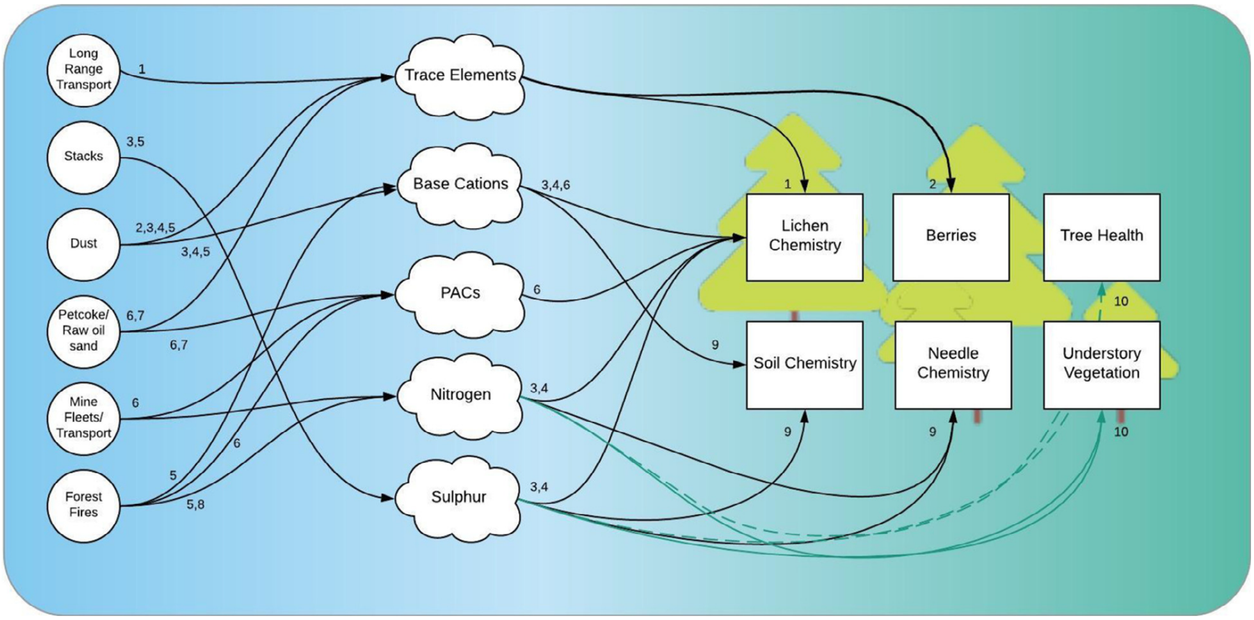

Fig. 4. Source apportionment results. Lines indicate pathways identified in this special issue. Green lines indicate effects suggestive of fertilization, and dotted green lines indicate potential effects (weak evidence) due to fertilization. 1: Graney et al. 2019 (lead). 2: Stachiw et al. (2019). 3: Landis et al. (2017). 4: Landis et al. (2019c) (lichens). 5: Landis et al. (2019a) (dust). 6: Landis et al. (2019a,b) (PAHs). 7: MacNaughton et al. (2019). 8: Edgerton et al. (2019). 9: MacKenzie et al. (2019). 10: Bartels et al. (2019).

sources. Lead from dust originated from oil sands ore (14%), dry tailings (10%) and haul roads (3%) (Graney et al., 2019). However, within 30 km of emissions sources, local sources dominate Pb deposition to lichens (Graney et al., 2019).

Stachiw et al. (2019) assessed whether trace elements on berries were from dust associated with weathering of rocks, or from a more highly processed source. The authors found that the dust contributing trace elements to berries is of crustal origin, and that dust levels were higher close to industrial operations, which suggests that though the dust may be of crustal origin, mining and oil sands industrial activities are contributing to higher loads (Stachiw et al., 2019).

Research supports the observation that dust is a major factor driving deposition of PACs, BC and trace elements. Meanwhile, S emissions from stack emissions and nitrogen from mine fleets, transportation, forest fires and stack emissions deposited to the surrounding forests, decline to background levels within 50 km. This complex combination of sources makes management difficult. While improvements to emissions controls led to demonstrable declines in ambient concentrations of SO_2, non-point source emissions of NO_2 and particulate matter continue to grow. Better analysis of temporal trends of deposition data will help clarify how management changes have altered deposition. However, evidence showing increases in N and the relatively recent challenge of petroleum coke dust suggests that challenges to forest health may continue and change over time.

6. How can the Forest Health Monitoring Program be improved?

In addition to the technical analyses found in the papers presented in this VSI, which address the technical elements of the FHM Program, an evaluation of successes, challenges and opportunities relating to program design and operation is now timely. Lindenmayer and Likens (2010) presented a framework describing the design and operating elements of a successful long-term monitoring program and identified elements subject to failure that compromise the efficacy and/or longevity of longer-term ecological monitoring initiatives (Table 1). We give a thorough analysis of the FHM program (Supplemental material) through the lens of their framework, and make detailed recommendations for program improvement (Table 1) but in general we found that the FHM program is question-driven, well designed to answer questions regarding the impacts of deposition throughout the AOSR, guided by a clear conceptual model (Foster et al., 2019), built on strong partnerships with industry, government, eNGOs and local Indigenous communities, and well integrated between assessment of stressors to effects. The program is now poised to adapt its conceptual model (Fig. 5) to consider new findings and implement important changes to ensure the integrity of the program long term.

That the FHM Program, developed with a conceptual model to detect acidification, has identified evidence of limited local risks of acidification occurring within about 50 km of emissions sources (Edgerton et al., 2019; MacKenzie and Dietrich, 2019) but also detected a possible fertilization effect, within the same radius (Bartels et al., 2019; MacKenzie and Dietrich, 2019) is a critical finding that speaks to the utility of the effects-based approach. Adaptive monitoring (Likens and Lindenmayer, 2018; Lindenmayer and Likens, 2010) advocates adapting monitoring programs whenever new information becomes available, therefore, expansion of the TEEM program conceptual model to include impacts from fertilization as suggested in Fig. 5 is warranted. The environmental indicators measured within the FHM Program have utility to not only detect both types of stress, they also permit discrimination between acidification, basification and fertilization. This is a particularly useful aspect of the monitoring program; other federal monitoring programs including the Canadian federal water Environmental Effects Monitoring Programs have identical indicators for pulp and paper effluent monitoring as for metal monitoring (fish condition, gonad size, liver size and length) which can differentiate between effects from fertilization and toxicity (Environment and Climate Change Canada, 2012). Work is required to differentiate the specific FHM Program end points for acidification, basification and fertilization (Table 2). The establishment of a comprehensive suite of triggers that correspond to endpoints listed in Table 2 with 20 years of monitoring data can provide robust

Table 1
Assessment of the Forest Health Monitoring (FHM) Program according to criteria in Lindenmayer and Likens (2010).

Lindenmayer and Likens (2010) monitoring program assessment criteria	FHM Program status	Successes	Challenges	Recommendations for improvement
Tier I – Type of monitoring program				
1. Curiosity-driven	Not a curiosity-driven monitoring program Program was designed with a clear conceptual model of potential impacts from acid deposition to forest health, and with the intent of supporting management actions	n/a	n/a	
2. Mandated	Has been referenced in historic regulatory approvals, current regulatory expectation of approval holders is restricted to funding support for monitoring activities	Historically strong government support and participation Data used in support of acid deposition assessment in a regulatory context	Understanding the scientific goals of the government agencies that manage program funding and oversee achievement of program objectives	1. Continue to build relationships under the new funding approach and seek opportunities to define regulatory mandates for the program 2. Develop linkages to other potential management tools aimed at managing fertilizing deposition
3. Question-driven	Broad objective statement, with specific questions regarding acid deposition levels and effects	Objective statement and questions remain pertinent Short-term studies guided by specific questions and protocols Reassessment of objective and questions after 10 years led to focused studies in support of FHM Program evolution	Occasional distraction from the objective and/or scientific questions Participant turnover led to degradation of understanding of the program objectives and questions	3. Consider a process to validate the objective statement, and validate and/or revise monitoring questions to address unexpected findings (fertilization, BC)
Tier II – Failure modes				
1. Lacking question(s)	Clear question for FHM Program Clear questions for supporting research and pilot studies	Objective statement and questions remain pertinent	Participant turnover led to degradation of understanding of the program objectives and questions	Recommendation 3
2. Poor design	Gradient approach based on distance from emission sources Sampling procedures defined and documented	Identified a zone of deposition for acidic, elemental and organic compounds within 50 km of emission sources Identified unexpected outcomes – fertilization responses and neutralization of acid input by BC deposition	Focusing on enhancing spatial coverage (increasing monitoring at larger distances) has resulted in an understanding that coverage within the zone of deposition may be inadequate Procedural changes through the period of monitoring have led to limitations in the ability to conduct some analyses Decisions designed to promote program efficiency did not harm the ability to assess deposition changes over space, but did harm ability to detect change over time	4. Examine options for increasing monitoring effort within the 50 km of emission sources, and assessing for other gaps in monitoring 5. Develop statistical approaches to better support detection of temporal changes related to deposition 6. Consider enhancements to the program to better understand fertilization and/or BC deposition & effects 7. Consider establishing procedures for obtaining statistical guidance whenever changes to the program are considered 8. Develop collaboration with groups modeling deposition to validate model and provide mathematical mechanism for testing hypotheses.
3. Laundry list (monitor everything)	Focused on effects-based sampling and data collection related that targeted acid deposition responses	Robust data collection allowed characterization of acid deposition level and ecological responses, while identifying unanticipated outcomes (fertilization, BC deposition)	In the early period of monitoring, soil, needle and lichen samples were often analyzed for all elements. This was thought to be an appropriate leveraging of sample collection costs, resulting in a database that was not manageable or useful for many elements	9. Consider evaluation of current sample and data collection protocols to ensure that fertilization and BC deposition and responses are adequately measured, while retaining the robustness of the acid deposition monitoring component
4. Inappropriate indicators	Focused on indicators that relate to acid deposition monitoring and ecological responses	Identified a zone of influence that bounds deposition of acidic, fertilizing, organic and other compounds	Maturation of the program needed to understand appropriate sampling locations	10. Evaluate the suite of indicators measured and samples taken in the context of value to current objective and questions 11. Support Indigenous-led development of forest health end points and limits of change

(continued on next page)

Table 1 (*continued*)

Lindenmayer and Likens (2010) monitoring program assessment criteria	FHM Program status	Successes	Challenges	Recommendations for improvement
5. Overly broad, restrictive implementation	Focused on measuring responses in acid-sensitive jack pine forest system Specific sampling locations and procedures outside of jack pine network as appropriate (e.g., lichen sampling)	Acidification possible within 50 km of emission sources, to the acid-sensitive jack pine system Sufficiently focused to ascertain acid deposition effects (within 50 km of sources), while being sufficiently broad to detect non-acidifying effects (fertilization, BC deposition)	Directly applicable to jack pine forests, may be missing effects in other systems (e.g., aspen, wetlands)	12. Examine potential for aspen monitoring, possibly at a reduced level of effort and within 50 km of sources, to determine breadth of ecological effects 13. Integrate with ongoing wetlands monitoring to design endpoints for fertilizing deposition in that ecosystem
Tier III – Success characteristics				
1. Good, clear, evolving question(s)	Clear objective statement, clear acid deposition question Source-to-sink approach adopted after 10 years of monitoring, retention of original objective Clear and focused questions in support of source-to-sink studies and pilot programs	Objective statement and acid input questions remain pertinent Source-to-sink evolution resulted in a significant advance in understanding of sources (source apportionment) Addition of jack pine forest edges bolstered original program design	Pressures to change objective statement and/or questions as participants changed, frustration that focus on single stressor (acid deposition) doesn't capture all potential stressor effects on forest health	Recommendation 3 and 7
2. Conceptual model	Clear conceptual model defined at program initiation	Conceptual model guided program operations for >20 years Was sufficiently robust to retain focus on key questions	Was only expressed in written format, not pictorially, which was a less than ideal communication approach	14. Update conceptual model to incorporate expected and unexpected findings
3. Good partnerships	Multi-stakeholder, consensus-based organization created good partnerships	Strong relationships among participants developed over 20 years	Participant turnover and realignment of funding processes have degraded some partnerships	15. Effort to reconstructing partnerships under the realigned funding regime will take effort, but will be valuable
4. Dedicated & strong leadership	Strong effective leadership now in place	Strong scientific and organizational leadership during the 6-year monitoring cycles	Short-term leadership typically on a contract basis, with the 5-year interval between major sampling and data collection campaigns lacking in consistent leadership	16. Continue progression away from a contract-based leadership process to one based on WBEA in-house leadership
5. Funding stability	Funding stability being regained after a recent period of transition from direct industry funding to indirect funding provided through a government process	Funding of >20 years of monitoring Funding of multiple short-term studies and pilot programs supportive of the monitoring program	Transition from direct industry funding and oversight driven by approval conditions to indirect industry support (funding) and regulatory management has resulted in short-term funding constraints	17. Continue to strengthen funding relationships, particularly with a vision to longer-term funding patterns (i.e., the 6-year sampling campaign)
6. Frequent data use	Frequency of data use increasing over the period of monitoring	Data used in evaluation of acid deposition effects as mandated under regulatory frameworks, used in impact assessments and project applications Data from shorter-term supportive studies and pilot programs used to understand chemical dispersion and ecological response processes	Fragmented data storage, sometimes difficult to access 6-year sampling cycle takes a long time to develop a useable data set for trend analyses	18. Initiate data consolidation procedures and integrate historical and currently-collected data in a single data structure to facilitate data access. Effort is ongoing. 19. Support integration of FHM Program data with larger Oil Sands Monitoring Program data structures.
7. Scientific productivity	20 years of FHM Program operations and findings published in this virtual special issue	Understanding of the zone of deposition arising from air emission sources in the AOSR Multiple publications from shorter-term studies, including reports designed to be of use to regulatory authorities	Long period of monitoring required to acquire sufficient data to determine trends	20. Continue to support publication and other communication materials for stakeholders and broader community. 21. Support integration with other Oil Sands Monitoring research programs.
8. Data integrity	Data consolidation and integrity was a weakness identified in the processes and analyses leading to this special issue	Increasing data consolidation and accessibility, increased trust in the data collected	Ensuring on-going attention to data integrity in a program with a 6-year periodicity of sampling and data collection	22. Continue data consolidation and improvements to accessibility, consider updating procedures for data management

Table 1 (*continued*)

Lindenmayer and Likens (2010) monitoring program assessment criteria	FHM Program status	Successes	Challenges	Recommendations for improvement
Tier IV – Impediments				
1. Organizational structure & retention of corporate memory	Some core personnel involved in program initiation in 1996 providing mentoring and continuity of knowledge Succession planning is underway, including knowledge transfer from contractors to WBEA staff	Participation of core individuals from the initiation of the program to present Improved processes for knowledge transfer, mentoring and training	Personnel turnover will continue, at both the oversight and technical levels	23. Continue with development and implementation of succession planning, knowledge transfer, mentoring and training programs 24. Consider enhancing the change management procedures, to ensure that all information underlying decisions is captured and available
2. IP rights	WBEA data are meant to be publicly available, reducing the impact of IP rights on data distribution	No restrictions on distribution of data or program reports No limitations imposed on publication of data in scientific papers	New funding procedures may affect the mechanisms by which data and reports are distributed, and scientific paper preparation and submission	25. Changes in funding procedures ongoing, continue work to align funding criteria with limiting or eliminating IP rights over data and reports 26. Potential inclusion of traditional knowledge will require development of new data sharing agreements and understanding of ethical use of these data.
3. Short-term success metrics	Short-term studies and pilot programs published, or data and recommendations included in internal reports	Multiple scientific papers from short-term studies	With a 6-year sampling cycle, a long period is required to obtain sufficient data for spatial and temporal analyses Participants requiring data and interpretations on a short-term basis can become frustrated and question the program	27. Consider defining short-term success metrics for outcomes supportive of scientific publication (e.g., data management, mentoring)
4. Funding stability	Funding provided annually, with longer-term workplans updated annually	Funding support provided for over 20 years	New funding procedures have resulted in delays in conducting some program components, with minor disruption	Recommendation 17
Tier V – Integration				
1. Balance between question-driven & mandated monitoring	A balance favouring the question-driven approach with data being provided for use in assessing conditions relative to regulatory frameworks has been developed	Data collected in 2004 showed that ecological response regulatory trigger levels due to acid deposition had not been reached Data collected in 2011–2013 indicates that acid-sensitive systems within a zone of deposition (within 50 km of emission sources) may have reached regulatory trigger levels	Retaining the appropriate balance	Recommendation 4, 14, 21
2. Multi-disciplinary Interaction	Continued integration among air, soil, vegetation and lichen specialists that was developed at the initiation of the program	Broadening of the specialists within these disciplines, plus recent interactions with specialists in other disciplines (e.g., aquatic chemistry)	Broadening of interactions among discipline specialists initially resulted in confusion, conflicting expectations of program activities, and communication of program objectives and questions	Recommendation 21
3. Linking question-driven & mandated monitoring	Balanced links have developed and persist	Acid deposition questions being directly addressed, with data available for use in assessing environmental condition against regulatory objectives and frameworks	Ecological data limited in a regulatory context to acid deposition, the focus of program questions and the issue for which regulatory triggers are available	Recommendation 3, 4

methods to assess change as well as the predictive capacity necessary to manage observed changes. These can guide discussions on the current state of environmental health and potential future trajectories with key stakeholders (Burger et al., 2008).

There are always new endpoints that can be measured, and including Indigenous perspectives on forest health would be a welcome way to define new monitoring endpoints. It is a much more difficult task to identify endpoints that can be removed. Soil sampling is labour intensive, and the chemical analyses are expensive. This has driven decisions to composite samples in the past. Once sufficient soils data, using current protocols that do not composite samples, have been collected, the program would benefit from a statistical analysis of soils data to determine whether the correct number of replicates are being taken to answer questions of changes over time and space, and if so, whether there can be cost savings realized by reducing replication.

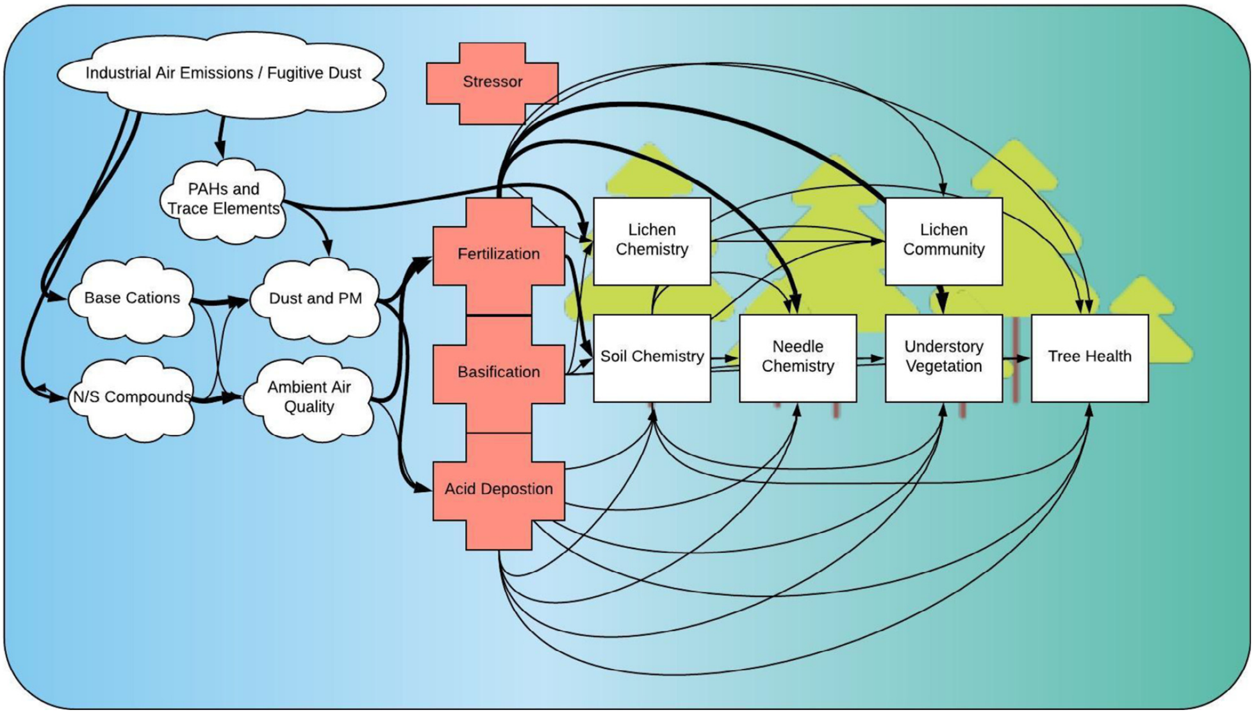

Fig. 5. Conceptual model of forest health monitoring, updated from Fig. 1 to reflect new information. Shown are three possible stressors; fertilization, acidification and basification, with potential effects. Relationships that have been demonstrated by papers in the Virtual Special Issue are in bold black lines.

The FHM Program chemical endpoints (IER data, passive data, soil and needle chemistry), particularly those for which there is dense temporal data, are candidate analytes for control charting (Arciszewski and Munkittrick, 2015; Arciszewski et al., 2018; Davidson and Spink, 2018) to establish triggers for focused monitoring or management of fertilization effects. This approach can also detect meaningful change in shorter timelines, including non-linear and non-monotonic changes, and provide a good basis to develop triggers to adapt monitoring (i.e. focused studies) where changes are identified (Arciszewski et al., 2017). This approach supports establishing adverse outcome pathways and affords greater capacity to attribute cause where impacts are observed (Ankley et al., 2010). Similarly, there is a need to establish triggers for the biological receptors (i.e. lichen community, needle growth/retention, vegetation and tree metrics) to initiate investigation of cause or management. Setting triggers for focused monitoring or management is not a purely scientific activity; it requires consideration of local values, broader perspectives of public interest, and tolerance of risk, and is therefore an activity outside the scope of the TEEM committee that requires government, industry, Indigenous and stakeholder collaboration.

Table 2
List of end points relevant to forest health monitoring and their applicability to reflecting sulphur and nitrogen deposition, fertilization or acidification effects.

Stressor	End point
S/N deposition	• Concentration of S and N in needles and soils (MacKenzie and Dietrich, 2019) • Increased concentration of S/N in lichens (Landis et al., 2019a)
Fertilization	• Increased tree growth (BAI) (Solberg et al., 2004; Bartels et al., 2019) • Increased understory richness (Thimonier et al., 1994; Bouwman et al., 2002; Bobbink et al., 2010; Bartels et al., 2019; Simkin et al., 2019) • Enrichment of N^{15} in soil pool (Laxton et al. 2010) • Enrichment of N^{15} in needles (Hemsley et al., 2019) • Shift to nitrophilous understory species (Bouwman et al., 2002) • Decreased C:N ratio in soil (Solberg et al., 2004)
Acidification	• Increased S in soil • Decreased BC in soil (CEMA, 2004) • Increased Al in soil (CEMA, 2004) • Decreased BC:Al in soil (CEMA, 2004) • Decreased base saturation in soil (CEMA, 2004) • Decreased pH in soil (CEMA, 2004) • Decreased needle retention (Driscoll et al., 2003) • Decreased tree resilience (increased disease, insect infestations) • Decreased tree growth (BAI) • Decreased lichen fitness (Gormanson et al., 2018) • Altered understory species community composition (Bobbink et al., 2010)

The initial set of FHM Program sites were chosen to include sites that represent high and low deposition according to modeled deposition levels, and sites added since program was initiated have focused on improving monitoring at larger distances from the emissions sources (Foster et al., 2019). This has resulted in relatively few sites between 10 and 50 km from emissions sources, the range over which deposition is exponentially declining (Edgerton et al., 2019). Adding sites that represent the mid-range of deposition would improve our understanding of the spatial extent of deposition and forest responses. An evaluation of the value of the larger number of distal sites, considering that oil sands development will continue to expand, is required to ensure that the program is efficiently designed and operated.

The FHM Program is also designed to support investigation of cause studies. With acidification and fertilization being inextricably linked, assessing patterns of receptor response with corresponding chemistry data is critical. Where multiple complex effluents have prompted the need for exposure characterization and effects monitoring in aquatic environments (Dube et al., 2005), attribution of cause has relied on the use of chemical tracers or indicators that are relevant to specific drivers of change (Hewitt et al., 2003). In the FHM Program, source apportionment studies have been key to identifying sources of deposition, and analyses that provide deposition estimates throughout the domain have been used to assess correlation of deposition variables with observed environmental change. However, a full understanding that supports a management response may require better understanding oil sands emissions from all sources identified in Fig. 4.

Understanding the challenges presented by complex stressors on the environment requires both modeling and effects-based monitoring, therefore reconciliation of findings from modeling, satellite measurements and deposition monitoring is needed. Modeling based on emissions and atmospheric processes is key for understanding cause and effect relationships, and the exercise of modeling is the only way to examine our assumptions about natural processes and our ability to measure change. Modeling provides region-wide predictions that may be prohibitive for monitoring programs and allows evaluation of development scenarios. At the same time, effects-based monitoring is necessary to ground-truth predictions, to assess the validity of model assumptions, and, most importantly, is the only method to detect unanticipated change. Differences between the predictions of the GEM-MACH model (Makar et al., 2018) and the findings of the FHM Program may stem from different approaches to critical loads, and difficulties in modeling BC deposition; similarly, the model suggests monitoring far downwind may improve the network. Collaboration between the two approaches could provide powerful insights into the deposition dynamics in and external to the AOSR.

The FHM Program has experienced difficulties managing long term data sets, documentation procedures for consistency and historical record, loss of sites to wildfire and development, and change management. With intensive forest campaigns occurring every six years, variances in procedures and their application have also occurred. As such, assembling and maintaining these data has been a challenge, and the importance of complete metadata, sample blanks and chain of custody practice cannot be overstated. The FHM Program has been fortunate to store biological samples for long periods; this practice is recommended though it is recognized that the costs and space requirements can be substantial. Data management for long term programs is an ongoing challenge (Lacarce et al. 2009), and considerable effort was expended for this VSI to collate, validate and store various types of forest health data collected since the late 1990s. The development of data standards that are provided to all contractors, and ongoing maintenance of databases, is crucial. As the FHM Program becomes more integrated with the Oil Sands Monitoring Program, there is an increased need to better integrate the FHM Program data with other programs (Swanson, 2019). In particular, the FHM Program would benefit from clearer integration with wetlands and acid sensitive lakes monitoring, which are also sensitive to acidifying and fertilizing deposition. Because many of the FHM Program ion exchange resins are placed under canopies to measure throughfall deposition (Edgerton et al., 2019), the predicted deposition based on them has limited utility to other ecosystems. However, there are also ion exchange resins placed in forest openings to measure bulk deposition, estimates that would likely apply to a range of boreal forest systems.

The FHM Program was initiated in no small part due to concerns expressed by Fort McKay First Nation about the potential impacts of atmospheric deposition on forest health. Indigenous communities have been key participants, with industry and government representatives, in the Terrestrial Ecological Effects Monitoring (TEEM) Committee that oversees the FHM Program. The importance of long-term relationships and collaborative program guidance cannot be overstated; stakeholder guidance and program review ensures it meets the needs of all parties, especially those who live in the region. This type of engagement is meaningful involvement: more than an invitation to be involved, it provides a structure for and involvement in decision-making (Gregory, 2000).

TEEM can now build on these partnerships to braid Indigenous perspectives into forest health monitoring. Traditional ecological knowledge is part of a worldview and way of knowing that has profound insights into the natural world (Cumulative Environmental Management Association and Group, 2015). Indigenous inclusion in environmental monitoring is becoming recognized as a best practice that benefits communities, the monitoring program and environmental management (Jollands and Harmsworth, 2007; Government of the Northwest Territories, 2012; Baker, 2016) along with a recognition that cultural resources and resources necessary to support the exercise of rights are both included in natural resources (Burger et al., 2008). A study on dust impacts on berries, berry quality and health, though not published as part of this special issue, was initiated and led by Fort McKay Elders in partnership with WBEA, and provides guidance on how monitoring programs can partner with Indigenous communities (Baker, 2016). Given that forest health is a significant concern of local indigenous communities, it is imperative to develop methods to include Indigenous perspectives, including developing new end points and limits of change that reflect the experience and ways of knowing of local Indigenous communities.

Finally, as the FHM Program becomes more integrated with other monitoring programs in the AOSR, it will become necessary to implement an adaptive monitoring approach, with tiers and triggers for intensive monitoring, to guide management. Rationalization of indicators through established temporal responses as presented in Fig. 2 provides a basis to assign additional monitoring intensity close to development, with less temporally and spatially intense monitoring for receptors that respond slowly over time. Adapting monitoring design to reflect indicator sensitivity with the support of established source attribution studies can provide a lens to direct investigation of cause activities where triggers are exceeded. Additionally, higher monitoring effort focused on indicators that respond more rapidly serves to protect against more latent irreversible responses (Fig. 2). Comparison of historical data across indicators can define trajectories to present accumulated state against which continued emissions can be used to predict adverse outcomes. This modeling of potential adverse outcomes is essential for future monitoring program adaptation, and enables calibrated predictions based on increased development within the region (Dube et al., 2013). Bartels et al. (2019) demonstrated that understory community structure and needle chemistry have proven to be indicators that respond to deposition and show change before N or acidification stress become apparent. These may be good candidates for to focus efforts in areas where deposition is having an effect. However, determining trigger levels and actions requires ongoing stakeholder engagement and is beyond the scope of this manuscript.

7. Conclusion

The FHM Program has completed 20 years of environmental monitoring in the AOSR, one of Canada's most intensively developed landscapes. The program, guided by a multi-stakeholder committee and a clear conceptual model linking emissions to acidifying deposition, has demonstrated some surprising changes in local forests. The FHM Program found evidence of acidifying deposition within 50 km of emission sources; however, due to the prevalence of BC deposition, the magnitude of acidification is mitigated. There is limited evidence of acidification effects in soils (MacKenzie and Dietrich, 2019) or on tree growth or understory composition; however, the FHM Program instead found evidence of fertilization effects on forest health endpoints (Bartels et al.,

2019) in areas closest to mining operations. Fertilization was attributed to increasing deposition of S and N close to emissions sources, concurrent with high deposition of BC (Edgerton et al., 2019; Landis et al., 2019c), which effectively neutralize acidifying deposition (Watmough et al., 2014; Fenn et al., 2015). The amount of N that can be deposited on a forest ecosystem is an area of ongoing research (Bobbink et al., 2010; Pardo et al., 2015; Simkin et al., 2019). Because the boreal forest in Northern Alberta does not have the same long-term history of forestry or industrial development as found in other monitored regions, and baseline levels of deposition can therefore be determined, research on critical loads for N deposition in the AOSR could provide needed input into understanding a global problem.

Despite the long-term nature of the program, detecting change over time remains difficult, though analysis of lichen concentrations suggests that S deposition increased between 2008 and 2014, and N deposition has held steady or decreased, with any increases coincident with the development of new facilities. Deposition of BC increased in magnitude and area between these time periods (Landis et al., 2019a). Source apportionment studies showed that dust is a major vector for deposition of BC (Landis et al., 2017), trace elements (Landis et al., 2017; Graney et al., 2019; Stachiw et al., 2019), and PACs (Landis et al., 2019a; McNaughton et al., 2019).

That BC deposition neutralizes acidifying deposition does not change the fact that considerable quantities of S and N are being deposited on a nitrogen-limited environment; thus, these remain stressors that affect the receiving environment. Further understanding of the dynamics of BC deposition and consideration of the potential effects of concomitant PAH and trace element deposition is required. Modeling that includes these dynamics will support the improved assessments of dust control efforts to reduce impacts.

An effective environmental monitoring program should also have triggers to focus attention on detected changes. Acidifying input greater than the 0.17 keq ha^{-1} $year^{-1}$ level that triggers investigative action under the provincial Acid Deposition Management Framework (Government of Alberta, 2008) has been estimated at some sites (Edgerton et al., 2019); investigative actions as described in that framework, including determining the PAI levels of whole grids and potentially MAGIC modeling may be appropriate within the next provincial assessment. Similarly, detailed assessments using the procedures in the regional Acid Deposition Management Framework (Cumulative Environmental Management Association, 2004; Whitfield and Watmough, 2015) may also be appropriate, focusing on the areas where deposition at or above 0.17 keq ha^{-1} $year^{-1}$ has been estimated. There are no monitoring or management triggers for fertilizing deposition. A 2008 draft framework for fertilizing deposition did not define critical loads for N deposition, due to the lack of an identified fertilization risk determined at the time and the lack of information to support this work (Cumulative Environmental Management Association, 2008). However, we suggest that the results of potential fertilization, the completion of key N addition studies (Watmough et al., 2016; Kwak et al., 2018; Wieder et al., 2019), the potential for increased NO_2 emissions as the industry expands, and the increasing relevance of NO_2 emissions to acidifying deposition all warrant source apportionment studies of N deposition and consideration of potential causes of any fertilization effect.

The FHM Program is a robust framework to find associations between deposition and forest health. It has been supported by an engaged multi-stakeholder committee that includes industry, government and Indigenous communities all of whom have guided all design and monitoring decisions, which has ensured the FHM Program's relevance to all stakeholders. Rationalization of sites and establishing change management procedures are required to ensure that the program remains rigorous and effective as the industry expands. The detection of BC and fertilization effects were unanticipated when the program was established; adaptive monitoring allows for the adjustment of monitoring questions and conceptual models in response to such new information (Lindenmayer and Likens, 2010; Likens and Lindenmayer, 2018; Arciszewski and Munkittrick, 2015). Further, the fact that the endpoints chosen by the FHM Program not only detected effects attributed to a different pathway of effect than originally conceived suggests that the FHM Program framework has the potential to inform our understanding of the effects of atmospheric deposition on forest health as the oil sands industry continues to grow, provided the conceptual model and approach are adapted in response to new information.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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