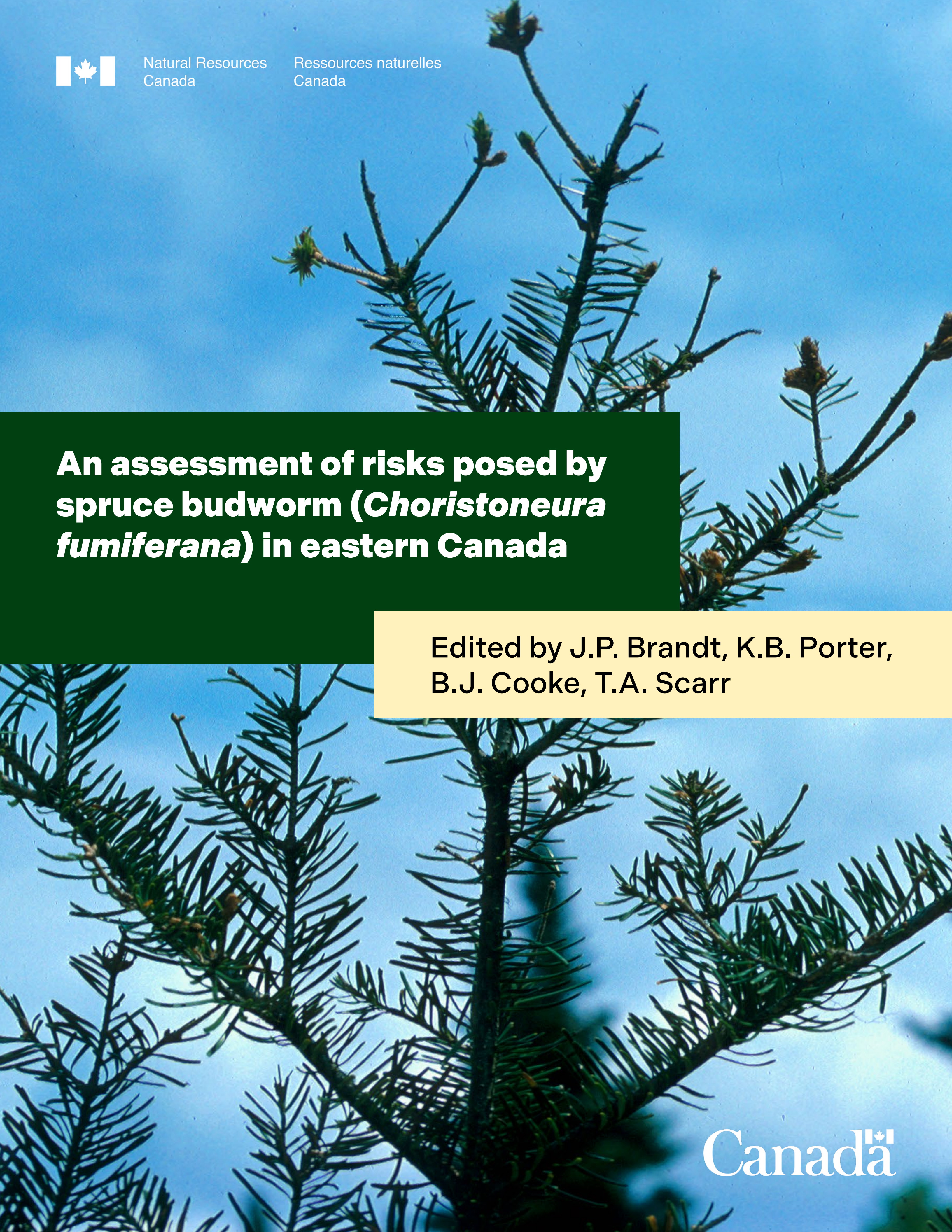

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An assessment of risks posed by spruce budworm (*Choristoneura fumiferana*) in eastern Canada

Edited by J.P. Brandt, K.B. Porter, B.J. Cooke, T.A. Scarr

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Introduction and Objectives

Spruce budworm (*Choristoneura fumiferana* [Clem.]; hereafter referred to as SBW) is a native insect that inhabits much of boreal and hemiboreal[1] North America east of the Rocky Mountains where its hosts are found (Harvey 1985; Nealis 2016). It is the most widely distributed and destructive forest insect of spruce-fir forests in Canada (Rose and Lindquist 1985; Hall and Moody 1994; Nealis 2016). Periodic outbreaks cause significant defoliation that leads to reduced photosynthetic capacity of trees, ultimately resulting in growth loss, top-kill, and tree mortality (Elliott 1960; Batzer 1973; Howse et al. 1980; Gross 1985). These impacts can lead to significant losses of important timber and non-timber resources, negatively affecting not only the forest industry and forest-dependent communities but also recreation, wildlife, and carbon sequestration (Prebble 1975; Dymond et al. 2010; Hennigar and MacLean 2010; Chang et al. 2012). The recurring outbreaks of SBW are a major natural disturbance in eastern North America and have important influences on the evolved adaptations of many native species (Venier and Holmes 2010).

Susceptibility and vulnerability[2] vary among host tree species and among forest stands. Both balsam fir (*Abies balsamea* (L.) Mill.) and white spruce (*Picea glauca* (Moench) Voss) are about equally susceptible to SBW (Mattson et al. 1991; Nealis and Régnière 2004). Red spruce (*Picea rubens* Sarg.) and black spruce (*Picea mariana* (Mill.)) are less susceptible. Such differences in susceptibility are best explained by synchronization of tree-insect phenology (Nealis and Régnière 2004; Hennigar et al. 2008; Régnière and Nealis 2018). The water and nitrogen content of the balsam fir, along with its needle softness, are at their maximum at the time of budburst, which is closely matched to the emergence of overwintering SBW second-instar (L2) larvae (Mattson et al. 1983, 1991; MacLean and MacKinnon 1997). White spruce is less synchronized with SBW, flushing about 4 days later than balsam fir (Greenbank 1963), and its nitrogen content is lower than balsam fir (Mattson et al. 1983). Red and black spruce are less suitable because budburst occurs about 13 days later than balsam fir (Greenbank 1963), meaning that there is no new succulent foliage for SBW larvae to feed on when they emerge on these hosts, which decreases larval survival (Blais 1957; Heron 1965; Hennigar et al. 2008). Balsam fir is also more vulnerable than any of the spruce because it carries a lower complement of foliage, and SBW larvae will feed on both new and older needles (Mattson 1985). Thus, the higher the proportion of balsam fir in a particular stand, the greater the impacts (Blum and Maclean 1985; MacLean and MacKinnon 1997).

As an economically important and damaging insect, SBW has been the subject of significant research for more than 100 years in Canada and in the US. The life cycle of the insect is well known and described in Rose and Lindquist (1985), U.S. Department of Agriculture (1985), and Johnson and Lyon (1988). During the 1970–1995 outbreak, the Canadian and US governments partnered and collaborated on the highly productive Canada-United States Spruce Budworms (CANUSA) Program, which resulted in many scientific and technical publications on SBW and western spruce budworm (*Choristoneura occidentalis* [Freeman]). Perhaps most notable were the reports titled *Spruce Budworms Handbook: Managing the Spruce Budworm in Eastern North America*, published in 1984 (Schmitt et al. 1984), and *Recent Advances in Spruce Budworms Research*, published in 1985 (Sanders et al. 1985). The latter report included more than 100 papers on various aspects of SBW and western spruce budworm biology and management.

[1]See Brandt (2009) for a definition of boreal and hemiboreal within North America.

[2]Susceptibility of a host or forest is defined as the probability that the host or forest will be attacked by SBW; vulnerability is the probability that damage, such as reduction in tree growth and tree mortality, will result from SBW defoliation (Blum and MacLean 1985).

Objectives

This report is intended to characterize risks posed by SBW outbreaks to a wide array of forest values (e.g., timber and non-timber products, trade and market access, fire risk, and other forest goods and services). This has been accomplished by (i) integrating our latest understanding of SBW biology, ecology, and management; (ii) qualifying our confidence in that understanding; (iii) identifying gaps; and (iv) identifying research priorities that will reduce uncertainty. The latter four elements were part of a multifaceted risk analysis (details provided below). Our work is not a systematic review but rather a synthesis of knowledge by experts, with support from the literature and external review. Ultimately, this report is intended to be a reference for forest pest managers in developing contemporary strategies for SBW management and to provide background for decision makers and policymakers whose goals are to minimize or mitigate risks associated with SBW outbreaks. The report is also intended to help research managers and researchers identify areas of inquiry that will help address knowledge gaps and reduce existing uncertainty.

Background Information

Past outbreaks

Records dating back several hundred years indicate SBW outbreaks have defoliated extensive areas of forests on a regular basis (Boulanger and Arseneault 2004; Boulanger et al. 2012; Simard et al. 2006) — every 30 to 40 years in eastern Canada (Boulanger and Arseneault 2004; Jardon et al. 2003). Outbreaks typically persist for 10 years or more. During the 20th century, eastern North America experienced three major SBW outbreaks with some regional variation: around 1910–1920, 1939–1958 and 1970–1995 (Kettela 1983; Hall et al. 1998). In Canada, the last major outbreak reached its peak in the 1970s, with moderate-to-severe defoliation affecting more than 50 million hectares of forest (see regional status reports in Armstrong and Ives 1995). Armstrong and Ives (1995) and the national reports of the Canadian Forest Service's former Forest Insect and Disease Survey (1970–1995; e.g., Sterner and Davidson 1981; Kondo and Taylor 1984) provide details, summarized below, on the 1970–1995 outbreak faced by each province in eastern Canada, as well as forest protection activities, most often in the form of aerial application of insecticides. Such activities, however, varied in magnitude and undoubtedly influenced the extent of defoliation and mortality estimates presented below on a province-by-province basis.

Nova Scotia recorded defoliation beginning in 1968. At the peak of this outbreak in 1976, moderate-to-severe defoliation affected 1.2 million ha on Cape Breton Island and in pockets in northern mainland counties. This outbreak collapsed in the late 1980s. The largest, most devastating outbreak of SBW ever recorded in the Maritimes occurred on Cape Breton Island during this time. Populations were extremely high, such that by the end of the outbreak only 10% of the balsam fir trees on the Cape Breton Highlands survived (and 6% in the lowlands).

New Brunswick also experienced an outbreak in the same time frame as Nova Scotia and peaked at 3.5 million ha of moderate-to-severe defoliation in the mid-1970s. Virtually all susceptible forests in the province were under severe attack during the 3-year period, 1973–75.

Spruce budworm populations in Prince Edward Island were also elevated with moderate-to-severe defoliation during much of the 1970s and into the early 1980s. Although most of this defoliation occurred in scattered patches, between 1975 and 1978 and again in 1981, widespread defoliation affected virtually all balsam fir and spruce stands across the island. A significant amount of balsam fir experienced repeated defoliation and died during this outbreak, and spruce trees, weakened by SBW defoliation, were then attacked and killed by spruce beetle (*Dendroctonus rufipennis* [Kirby]).

Spruce budworm also became a major economic forest insect in Newfoundland and Labrador when some defoliation first became apparent in 1971 and lasted until about 1988. The size of the infestation, the intensity of defoliation, and the degree of tree mortality were historically unprecedented on the island of Newfoundland. Prior to 1970, only a few relatively localized infestations had been recorded, lasting only a few years, and causing no noticeable tree mortality. Between 1971 and 1974, the infestation spread across the island from west to east. Feeding intensity by the insect on balsam fir and white spruce became extreme from 1976 to 1978, causing 1.3 million ha of moderate-to-severe defoliation. The resulting mortality of balsam fir and spruce during the outbreak represented 15% of the island's softwood growing stock.

Spruce budworm is the most damaging insect in Quebec's conifer forests. The first surveys to report defoliation during the previous outbreak were in 1968. Like the three outbreaks earlier in the 20th century, it began in western Quebec. Large patches of defoliation appeared in the Lower St. Lawrence region in 1970 and in the Gaspé Peninsula in 1972. The infestation reached its peak in 1975 when areas of moderate defoliation covered more than 28 million hectares. By the mid-1980s, the outbreak had collapsed after a total of 12.9 million hectares of tree mortality, primarily involving balsam fir.

Spruce budworm is also the most destructive forest insect in Ontario. The previous outbreak began in 1967 and continued until 1996. Three spatially discrete, regional outbreaks developed in the late 1960s that fell into the broad geographical areas of northwestern, Lake Nipigon, and northeastern Ontario (Candau et al. 1998). Overall, budworm infestations irrupted and spread quickly from the late 1960s to 1980, when the outbreak peaked at 18.8

million ha. After the peak, total outbreak area declined to 8.7 million ha in 1984, increased to 12.3 million ha in 1985, but then declined further in 1986 and 1987 to 7.2 million ha. The gross area of tree mortality totalled slightly more than 14 million ha in 1987.

Past management

Considerable resources have been devoted to SBW management. Prebble (1975) published a thorough review of the history of research on products and technology and on the operational management of SBW and other forest insects, for the period of 1927 to 1973. Further research on SBW and other forest insects during the period 1973 to 1988 is reported by Armstrong and Ives (1995). Here, we provide a brief summary for context: Attempts at management began in the early part of the 20th century. Much of the initial work focused on parasites and predators (McGugan and Coppel 1962; Wallace 1995). Chemical insecticides were applied aerially on a trial basis in Ontario and Nova Scotia beginning in the late 1920s, but it was not until after WWII that chemical insecticides (DDT, a persistent organochloride) began to be applied operationally across extensive areas of forest by aircraft (Nigam 1975). By the 1960s, the negative environmental consequences of DDT led to a switch to non-persistent organophosphates and carbamates (Nigam 1975). During the same period and continuing into the 1970s, research was completed on the development and testing of the bacterium, *Bacillus thuringiensis* (*Bt*), as an insecticide (Cameron 1975; Morris et al. 1975), as well as insect growth regulators (Outram 1975; Retnakaran 1995), the latter of which culminated in the use of tebufenozide (Sundaram et al. 1996). As insecticides and spray technology evolved, and as scientific research contributed to a greater understanding of both SBW population dynamics and environmental impacts of various control products and approaches, the various strategies employed against SBW adapted as well. The latter strategies included population control, foliage protection, and, finally, pre-emptive or proactive control or early intervention (Prebble 1975; Baskerville 1976; Kettela 1995a, 1995b; Hartling 2000; Johns et al. 2019). Researchers also examined the effectiveness of some of these strategies by targeting different life stages of the insect. Initial strategies focused unsuccessfully on population control by targeting feeding larvae (Miller and Kettela 1975) or moths (Kettela 1995a). Later, and continuing to the present, foliage protection was implemented across much smaller areas by applying insecticides to high-value host stands only and targeting feeding larvae. A "foliage protection strategy" (FPS) is considered reactive and focuses on keeping defoliated trees alive until the outbreak subsides or trees are harvested. A pre-emptive, proactive, or early intervention strategy (EIS) is designed to suppress low-level populations before they build to an outbreak and cause significant defoliation. Although the latter strategy has been attempted on several occasions in Ontario, Quebec, and Alberta (as reviewed in Hartling 2000), it was not until more recent refinements in our understanding of populations dynamics that the present form of EIS has taken shape. The early intervention strategy in its present experimental form incorporates the extensive surveillance of a high number of monitoring plots, and early detection in areas with low-level but increasing populations that are then targeted by application of insecticides (based on either *Bt* or tebufenozide) to prevent them from exceeding the irruption threshold that leads to outbreaks (Johns et al. 2019; MacLean et al. 2019). The current EIS attempts to maximize benefits and minimize any potential negative consequences by preventing outbreaks, but its applicability to broader areas where SBW outbreaks occur remains uncertain.

Current outbreak status and management

The current eastern Canadian SBW outbreak began in the Côte-Nord region of Quebec in 2006, which was the first year since 1991 that more than 50,000 ha were defoliated. As of 2021, much of the current outbreak is still centred in Quebec, but populations of the insect have recently risen in the past few years in Newfoundland and in Ontario (Fig. 1). Populations in New Brunswick have thus far been kept in check through the experimental implementation of the EIS since 2014, and the same approach is being attempted in western Newfoundland. Spruce budworm populations in Newfoundland, however, are augmented periodically by large influxes of immigrating moths resulting from the long-distance dispersal from Quebec, and the unique biogeography of the island may present a challenge to the management of SBW in that province. The applicability of the EIS beyond the end of the current experiment in 2026, and to broader areas where SBW outbreaks occur, is untested, and there are likely key questions that will remain unanswered at the end of the experiment. Both Ontario (since 2020) and Quebec (since 2009) are using the more conventional FPS. In 2021, Quebec treated 742,000 ha of forest affected by SBW in five regions. In northeastern Ontario in 2020, the province implemented an FPS for the first time since 1991, treating more than 55,000 ha of affected forest. Ontario treated just under 54,000 ha in 2021. Ontario's treatment program is expected to continue and increase in size during the next few years as the SBW outbreak is likely to grow. Spruce budworm defoliation has also occurred in US states adjacent to Canada: Minnesota, where 150,000 ha were defoliated in 2021, and Maine, where less than 1,000 ha were defoliated in the same year.

Although the aerial application of insecticides has been an important method of managing SBW outbreaks since the 1950s, forestry jurisdictions in Canada have been implementing a variety of management options when trying to minimize the negative impacts of a SBW outbreak (e.g., Cuff and Walker 1985). The options and their application have continued to evolve as forest management paradigms have changed. Beyond FPS and EIS, silvicultural approaches afford another set of options for forest managers when dealing with a SBW outbreak. These approaches mitigate the impacts of outbreaks locally but do not reduce the broader risks. They include (i) pre-emptive harvesting and accelerated harvesting of host stands expected to be defoliated and otherwise damaged by SBW, and (ii) salvage logging, which is the harvesting of trees that are dead, dying, or deteriorating before the fibre of the affected trees loses its economic value. Another option available to managers includes no action, where the outbreak is left to run its course without interventions. The choice of any management option depends largely on what the specific forest management objectives are for any given stand or area.

Risk Analysis Process and Scope

Risk is the product of the likelihood of occurrence of an event and the expected consequences of its occurrence. Risk analysis can be broadly defined to include risk assessment, risk response, and risk communication (Hodge 2015; Nealis 2015). Risk assessment and response are knowledge-based facets of the analysis; they answer the questions of “what do we know?”, “what does it mean?” and “what should we do?” (Nealis 2015). In a forest pest management context, risk assessment is an analysis of the probability of an event occurring (e.g., a population outbreak of a native species, or the introduction of a non-native species), the spread and extent of that event (e.g., what geographic area could be affected), and the impact of that event on forest values (e.g., timber and non-timber products, trade and market access, fire risk, and other forest benefits). Risk response is an evaluation of the management options (e.g., silviculture, direct population control, and regulatory actions). Risk communication conveys information upwards within natural resource management agencies to decision makers, downwards within the organization to those implementing the response, and outwards to partners, collaborators, stakeholders, and Indigenous peoples.

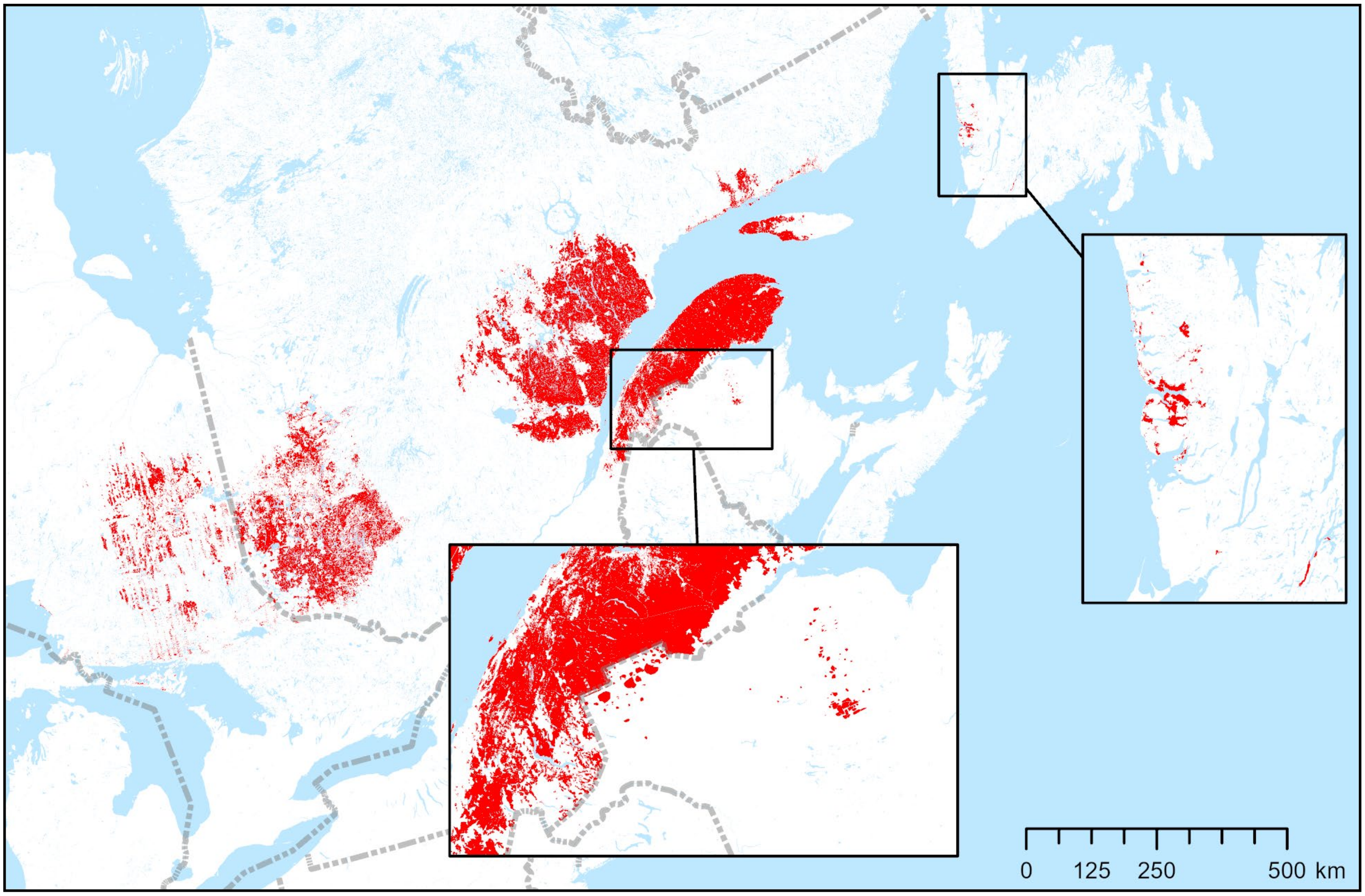

Figure 1. Extent of spruce budworm defoliation in eastern Canada in 2021

The risk assessment described in this report follows the Risk Analysis Framework developed under the Forest Pest Working Group of the Canadian Council of Forest Ministers in support of the National Forest Pest Strategy (Hodge 2015; Nealis 2015). Provocative affirmative statements are used in a workshop of scientists and forest pest management experts to elicit discussions around scientific evidence and expert opinion to synthesize the relevant knowledge and identify gaps. For each statement or topic, associated uncertainties and the research needed to reduce the level of uncertainty are identified. Ultimately, this evidence and uncertainty are integrated to determine what it means with respect to a particular pest risk. It is an integrative and adaptive approach where new knowledge, uncertainty and appropriate responses are repeatedly assessed as new information becomes available. An earlier risk assessment for SBW was completed in 2011 (Nealis 2011).

The current state of SBW outbreaks and increasing populations in eastern Canada along with recently published and unpublished scientific information, the testing of potential new management approaches, and subsequent new research needs have triggered Natural Resources Canada to undertake an updated analysis of risk. The overall goal of this risk assessment is to conduct an analysis of the risk posed by SBW outbreaks in eastern Canada (Ontario and eastward) during the next 10–15 years, and the effects of managing the insect. The analysis articulates the differences across jurisdictions in the forest sector, the values at risk from outbreaks and management, and the current understanding of SBW ecology.

Research scientists and experts from the forest pest management community were invited to participate in four 3-hour workshop sessions facilitated by Natural Resources Canada staff. The workshop included a discussion of the evidence, uncertainty, and ways to reduce uncertainty for several SBW topics. Following the workshop, participating researchers and pest managers were asked to form writing teams to prepare sections of the risk assessment report. Using notes captured during the workshop and incorporating additional published and unpublished information and expert knowledge to fill gaps, teams were tasked with completing a synthesis of their assigned topic. Topics are based on one or more affirmative statements (AS), and syntheses are informed by a discussion of the evidence to support or refute the statement, followed by ratings of uncertainty[3] for knowledge pertinent to the topic. Affirmative statements 1 to 3 are on population dynamics and SBW outbreaks. Affirmative statement 4 is on the expected consequences of climate change on SBW. Affirmative statement 5 is on the interaction between SBW outbreaks and fire. Affirmative statements 6 to 8 are on the impacts of outbreaks. Affirmative statements 9 to 14 are on various aspects of SBW management, including monitoring, EIS, FPS, pre-emptive harvest, and salvage logging. Affirmative statement 15 is on the impacts of SBW management on non-target species. Please note that there is necessarily some overlap between AS because writing teams worked independently and because of linkages between subject matter in different AS.

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[3]Low uncertainty indicates that the supporting evidence and scientific data are locally applicable, consistent and comprehensive, and any expected variability will not change the validity of the statement or assertion.

Moderate uncertainty indicates that either (a) the statement/assertion is supported by preliminary evidence that could significantly lower the uncertainty, or (b) there is inherent variability that could significantly change the magnitude of the statement/assertion but not its truth.

High uncertainty indicates that supporting evidence and scientific data are missing, are not locally applicable, and/or are inconsistent, and the expected variability could change the validity of the statement/assertion.

What We Know and Do Not Know About SBW

Population dynamics and outbreaks

AS 1: SBW outbreaks are periodic in occurrence, and the mechanisms of population irruption are well understood (By B.J. Cooke, J. Régnière, J.-N. Candau)

Evidence

Key points:

- Extensive periodic outbreaks have been observed across several centuries within the dendroecological record, and over several millennia within the paleoecological record.
- There is significant spatial variation in outbreak behaviour, some of which appears to be attributable to spatial variations in climate and forest structure, although much of the variation in periodicity, intensity and duration remains unexplained.
- The phenomenon of epicentre formation is not a mere historical anomaly but was a prominent feature in the early development of the current outbreak cycle in northern Quebec. The early phase of population cycle initiation is marked by a period of positive density-dependent recruitment caused by improved mating success with rising density through time.
- Although severe budworm outbreaks do result in tree mortality, in the bulk of the insect's range, the outbreak cycle terminates with only growth loss and minimal tree mortality.
- A wide range of natural enemies induce periodic population cycling. Spruce budworm-natural enemy interactions are not simple, however, and are likely mediated by environmental factors, including forest composition and its influence on not just the community of natural enemies affecting budworm but also their own predators and their own suite of alternate hosts.

Details:

Spruce budworm is an integral part of boreal and hemiboreal forest ecology across most of the range of its hosts, with extensive outbreaks observed across several centuries within the dendroecological record (Boulanger and Arseneault 2004; Boulanger et al. 2012) and over several millennia within the paleoecological record (Simard et al. 2006). The long-term persistence of SBW and coexistence with its primary host tree species suggests a certain degree of stability and resilience at the ecosystem level, even if there exists a level of destructiveness and economic impact, sometimes intolerably so, at the level of stands or watersheds (Chang et al. 2012b) (see section on Impacts of outbreaks, AS 6 and AS 7).

The most reported outbreak interval is in the order of 30–40 years (e.g., Jardon et al. 2003), and the insect is well known for regionally synchronized outbreaks (Royama 1984; Peltonen et al. 2002) that cause widespread decline of hosts across broad areas (MacLean 1980). However, a wide range of outbreak frequencies and spatial scales of synchronization has been observed in different parts of the insect's extensive range (e.g., Williams and Liebhold 2000; Robert et al. 2012).

Despite the quasi-periodic recurrence of outbreaks, forecasting is a challenge, as there is significant spatial variation in outbreak behaviour (Candau et al. 1998; Gray and MacKinnon 2006). Some of the spatial variation in outbreak intensity appears to be attributable to spatial variations in climate and forest structure (Candau and Fleming 2005; Gray 2008, 2013). Much of the variation in periodicity, intensity, and duration, however, appears to have no clear explanation (Robert et al. 2018; Cooke et al. 2024).

The mechanisms underlying the initiation and termination of outbreak cycles have been the target of population ecology research for decades. Spruce budworm cycles slowly, such that population data accrue slowly. Sturtevant et al. (2015) comprehensively reviewed 15 different forest-budworm outbreak models and traced the evolution of thought regarding the causes of the SBW outbreak cycle and its deviations from strictly periodic behaviour.

Hardy et al. (1983) emphasized the importance of focal epicentres as early sources of large-scale outbreaks. The same process was suggested by Candau et al. (1998) to explain the patterns of defoliation in Ontario during the last outbreak. Bouchard and Auger (2014) illustrated that the phenomenon of epicentre formation was not a mere historical anomaly but was a prominent feature in the early development of the current outbreak cycle in northern Quebec. Pureswaran et al. (2016), in a comprehensive review, showed that the early phase of population cycle initiation is marked by a period of positive density-dependent recruitment caused by improved mating success with rising density through time—a clear source of irruptive behaviour over and above the oft-cited slow-cycling behaviour. This process may well explain the historical pattern whereby outbreaks are initiated from focal epicentres.

One might expect that the collapse of the SBW population cycle is necessitated by the collapse of the forest. In the bulk of the insect's range, however, the outbreak cycle terminates without large-scale tree mortality. The evidence for what causes the termination is largely anecdotal and, therefore, should be supported through additional observation. In favour of it are the following fragmentary observations:

1. Studies of outbreak defoliation patterns in Quebec suggest that there are three main modalities of outbreak: light, moderate, and severe (Gray et al. 2000). Something other than stand mortality must be responsible for terminating the outbreak cycle in those stands that only receive light defoliation for a short duration. A limitation here is that studies of defoliation patterns lack any data associated with survivorship.
2. Tree-ring reconstructions of historic SBW outbreaks are conducted using stems that survive outbreaks. If there were no survivors of outbreaks, there would be no tree-ring studies. Stands with only partial mortality are specifically sought out for sampling, as an ideal source of abundant historical outbreak data. There is not a region in Canada that has not been targeted for tree-ring reconstructions of SBW outbreaks. In Quebec, Navarro et al. (2018) suggested that the last three outbreak cycles of SBW through the 20th century affected 40%, 30%, and then 50–70% of stems. Here "affected" refers to a significant sharp and sustained growth loss in surviving stems. More than a thousand kilometres to the west, in Minnesota and northwestern Ontario, Robert et al. (2018) found an identical pattern. Roughly half of the surviving stems in any given outbreak cycle show no obvious sign of heavy defoliation. This implies outbreaks in general must be terminated by something other than stand liquidation. Based on the same three outbreak cycles of the 20th century, Berguet et al. (2021) showed that in one third of Quebec the outbreak cycle will last significantly shorter than in the other two thirds. Outbreaks here are not terminating from stand depletion.
3. Spruce is not nearly as vulnerable as balsam fir to mortality from defoliation (Blais 1981; MacLean et al. 1984; Ostaff and MacLean 1989), and the pure white spruce stands of eastern and western Canada survive better than the pure fir stands of New Brunswick. Outbreaks in these western spruce stands frequently terminate without heavy mortality. Red spruce (whose distribution is limited to eastern North America) and white spruce live to hundreds of years in age, and, by virtue of their resilience, are the preferred species for reconstructing SBW outbreaks. As an extreme example, in the western spruce budworm system, the latter budworm rarely kills its host trees, which are the extremely long-lived Douglas-fir (*Pseudotsuga menziesii*). Outbreaks of western spruce budworm cycle persistently, attacking the same tree every few decades for centuries (Swetnam and Lynch 1989). It is reasonable to posit that the same factors affecting western spruce budworm are involved in SBW (Nealis 2016). This is not definitive proof, but it is suggestive evidence.
4. The fact that outbreaks in white spruce stands and landscapes often terminate before stand depletion does not imply that the terminating factor is not dependent on foliage availability; it could be. However, it remains to be demonstrated that heavy defoliation or partial mortality alone (e.g., less than 50%) might be capable of triggering some hitherto unidentified mortality or emigration factor. Quantifying this issue remains a valid research objective.

Royama (1984) argued that a wide range of natural enemies is both necessary and sufficient to induce periodic population behaviour. He also cautioned that the SBW-natural enemy interaction was not simple, but was likely to be mediated by environmental factors, including forest composition and its influence on not just the community of natural enemies affecting budworm but also their own predators and their own suite of alternate hosts—a proposition that is supported by long-term studies of budworm food web ecology (Eveleigh et al. 2007).

The bulk of SBW research was conducted through an era of relatively stable climate, from the 1950s through the early 2000s. A common theme throughout this period was the role of harsh winter and spring weather in producing low survival rates affecting cycling properties in the short term. As climate has warmed appreciably through the last two decades, studies have increasingly identified climate as a key factor in driving a northward range shift in SBW activity (Candau and Fleming 2011; Régnière et al. 2012; Berguet et al. 2021) (see also section on Consequences of climate change, AS 4).

Additionally, SBW population and food web monitoring is expensive, and so there is a hard practical limit to our ability to make quantitative predictions of impact in the 10–15-year time horizon. Too much in the SBW food web can change too quickly to say much beyond what might happen after a year or two. In a complex system like that of SBW where there are many interacting parts, each vulnerable to disruption, uncertainty cascades through the system because of the many interdependencies (Dukes et al. 2009; Boulanger et al. 2016).

Uncertainty

1. The overall level of uncertainty on SBW population dynamics is low relative to the needs of forest managers pursuing an FPS. This low level of knowledge uncertainty, however, does not equate to a low level of predictive uncertainty in the forecasting context. The moderately high level of forecasting uncertainty that most forest managers would find tolerable becomes intolerable when compared to some of the finer demands of forest managers pursuing outbreak prevention policies. These managers will want to know with fair precision at what density the unstable outbreak threshold occurs. They will also want to know how the threshold evolves through time and space and as a product of ecological and climatic circumstances. There is moderately high uncertainty on these functions and parameters.
2. Moderately high uncertainty on how climate change could impact specific classes of natural enemies (specialists, semi-specialists, semi-generalists, and generalists), resulting in non-stationary disequilibrium dynamics regardless of their perceived role in cycling through a historically stable climate.
3. Moderately high uncertainty on the effects of changes in forest composition, both natural and anthropogenic, on natural enemy communities. This would affect predictions of future outbreak behaviour in response to changes in forest structure resulting from salvage logging or wildfire, for example (see AS5 and AS14). This uncertainty is compounded by the mysterious presences and absences of natural enemies that alternately seem to be associated with populations that either refuse to outbreak or collapse.
4. Low uncertainty that SBW outbreaks recur periodically.
5. Low uncertainty that outbreaks tend to start in focal epicentres, and yet there is a low capacity to predict how quickly populations will grow and spread from these epicentres and where they may be located in a future outbreak.
6. Low uncertainty that outbreak cycles tend to cover large areas as they grow during a decade or two, and yet there is a low capacity to predict how extensive any one outbreak may become.
7. Low uncertainty as to what causes SBW population irruptions and cycling, and yet forecasting the growth and spread of specific outbreaks is restrained by limited data about the status of the SBW food web through space and time. Prediction uncertainty grows relatively quickly the longer the desired time horizon of prediction.

Reducing uncertainty

1. Establish well-resolved studies that are appropriately rooted in the SBW ecology literature, underpinned by specific hypotheses of the mechanisms by which changes in forest structure and climate in time and space are likely to influence irruption and cycling dynamics. The single biggest mystery of SBW outbreaks continues to be the fundamental cause of the population "cycle" and inexplicably large variations in cycle amplitude.
2. Establish studies that examine basic SBW population ecology and community ecology. Although so-called top-down forces of natural enemies may play a critical role in cycling, so-called bottom-up forces, such as plant flowering and weather, may play a role in shaping cycling behaviour (Bouchard et al. 2018a).
3. Determine the mechanisms of population irruption for endemic populations. Rather than terminating research when budworm goes endemic, the opposite should happen.

4. Conduct synthetic spatial modelling to develop policy scenarios that can help with long-term forest management planning, including silvicultural approaches to damage mitigation. Spatial epidemiological modelling can also help develop more comprehensive hypotheses regarding the spatial scales across which irruptions are and are not preventable.

AS 2: SBW population irruptions are preventable (By B.J. Cooke, J. Régnière, J.-N. Candau)

Evidence

Key points:

- The central thesis of the early intervention approach to population management is that there is an unstable low-density population irruption threshold below which growing populations may be driven by insecticide application. Recent data suggest that the irruption threshold exists, is unstable, and occurs at roughly three to five L4 larvae per branch or a comparable population level of seven L2 per branch.
- If the irruption threshold drifts because of forest growth through time, then the question of whether prevention is feasible becomes more challenging to answer. Prevention may be possible in the short run, but this result may eventually become infeasible as the threshold is exceeded. Similarly, if there is a propensity for cycling induced by natural enemy interactions, then preventability may wane as natural enemies become increasingly scarce, resulting in an insurmountable upward pressure on budworm population growth rates.
- The emergence of the current SBW outbreak in Quebec has afforded the opportunity to gather new data on the role of natural enemies in releasing populations from endemic levels. So far, there is no evidence of an impoverished natural enemy community, despite there having been two decades of low budworm densities between the collapse of the last outbreak and the initiation of the current one in 2006. The EIS in New Brunswick has been successful in limiting population growth across the entire province, and there is no evidence of natural enemies dropping out and thereby releasing their regulating effect on budworm populations.
- The preventability of budworm population irruption is a practical matter of scale and economics. An irruption that is preventable at the level of the tree, stand or focal epicentre may not be preventable at the level of the landscape, because there may be economic, regulatory, or logistical barriers to upscaling an intervention program. Prevention requires continuous early detection, and the challenging logistics and economics of early detection can be a formidable barrier to pursuing the EIS. What is feasible for a small, highly timber-dependent province may be practically and economically infeasible for larger provinces.
- Localized population irruptions at the level of the tree, stand or watershed may be indefinitely preventable. Larger-scale, region-wide outbreaks may be preventable for a certain amount of time, but it remains to be seen whether the approach can be applied long-term. At the largest scales, cycling behaviour may eclipse all locally irruptive behaviour.

Details:

Johns et al. (2019) outlined an ecological framework for evaluating the feasibility of a range of SBW outbreak management strategies. At one end of the spectrum are forest managers who can tolerate some timber losses and are interested primarily in mitigating the worst of the damage expected during the course of a 20-year outbreak cycle. Commonly referred to as a "foliage protection strategy" (FPS), it is a "reactive" strategy, where intervention occurs only after evidence indicates risk tolerance thresholds are likely to be exceeded. Here, intervention is limited to just those areas where the risk of losses (most often economic) is imminent. At the other end of the spectrum, where risk tolerance is far lower, lies the proactive "early intervention strategy" (EIS) of aggressive early detection and population suppression before an irruption occurs, with its attendant risks of population growth and spread.

The central thesis of the EIS to population management is that there is an unstable low-density population irruption threshold below which growing populations may be driven, by intervention (e.g., insecticide application), if they ever exceed that irruption threshold (Johns et al. 2019). Should local populations be dropped below that threshold, they may proceed to either local extinction or local endemic, depending on the existence of a stable lower-density endemic equilibrium state. In this framework, the process of irruption is "fast", occurring on the timescale of just a few years, and the response is early or rapid, either maintaining a population below the irruption threshold, or pushing it back below the irruption threshold the instant it is exceeded.

Recent data suggest that the irruption threshold is unstable and that it occurs at roughly three to five L4 larvae per branch (Régnière et al. 2019a) or a comparable population level of seven L2 per branch (Johns et al. 2019). In the modern concept, the irruptive growth process is not driven by lack of predator pressure alone, as imagined by Ludwig et al. (1978), but also by enhanced mating success, as discussed by Pureswaran et al. (2016). This process, documented by Régnière et al. (2013), is thought to also be influenced by tree height and age, through foliage density—which aligns roughly with the vision of Holling—such that population aggregations leading to irruptions and epicentres are more likely to occur in overmature white spruce stands, including relic stands that survived past budworm outbreaks.

If the unstable irruption threshold exists and is static, then outbreaks are, in theory, preventable. If, however, the irruption threshold drifts because of, say, forest growth through time, then the question is more challenging to answer. Prevention may be possible in the short run, but this result may eventually become infeasible as the threshold is exceeded. Similarly, if there is a propensity for cycling induced by natural enemy interactions, then preventability may wane as natural enemies become increasingly scarce, resulting in an insurmountable upward pressure on budworm population growth rates. A critical question is, therefore, whether that upward pressure is, in fact, insurmountable, or at least under what conditions.

There are two lines of evidence that delayed feedback from specialist natural enemies must play some part in the outbreak cycle, even if it is limited. The first is the periodicity of irruptions. Although irruptions vary in frequency and amplitude in space and in time, the periodicity is robust enough that every large-scale, long-term study reports a similar result: outbreaks occur every 26–40 years. The second is the fact that budworm outbreaks often terminate before destroying all hosts in the stand. Something other than host availability must be responsible for those terminations.

The review by Sturtevant et al. (2015) documents the evolution in thought about SBW population dynamics and discusses the data underlying the tide shift in thinking from the 1970s to the present. Following the definitive monographs by Royama (1984, 1992), the bulk of the evidence supported the idea of robust population cycling—a scenario in which there is no irruption. There is only a cycle, whose momentum is so strong that little can be done to sway the population's cyclic trajectory. This interpretation represented a revolutionary departure from the earlier Ludwig et al. (1978) proposition, which held that the system was not prone to natural enemy-driven cycling, but rather that all outbreaks begin and end with the forest cycle.

However, the emergence of a new SBW outbreak in Quebec beginning in the 2000s has afforded the opportunity to gather new data on the role of natural enemies in releasing populations from endemic levels. So far, there is no evidence of an impoverishment of the natural enemy community, despite two decades of low budworm densities (Bouchard et al. 2018b). Meanwhile, the EIS in New Brunswick has been successful in limiting population growth at the scale of the entire province, with no sign of natural enemies dropping out and thereby releasing their grip on budworm populations. Régnière and Nealis (2019a) have suggested that the role of natural enemies is not nearly as strong as suggested by Royama (1984). According to the interpretation of Régnière and Nealis, if natural enemies are playing a role in generating the sort of lagged density-dependent feedback that induces cycling in simple theoretical models, that role is weakened by forces yet to be explained. This could include, for example, a significant role played by the rich hyperparasite community that attacks budworm parasitoids (Eveleigh et al. 2007). This is an ongoing area of active research.

Today, it is clear that both the interpretations of Ludwig et al. (1978) and Royama (1984) are unsuitable, and that the truth may lie somewhere in the middle, as a hybrid proposition (Sturtevant et al. 2015). The more data we accrue in the 21st century, the more strongly the consensus shifts back to the idea that the irruptive part of the outbreak process may dominate the cyclic component. If the new emerging consensus is correct, it supports the idea that the irruptions that dominate the outbreak cycle may be entirely preventable.

Two additional lines of evidence from other systems help frame this thinking. First, the irruptive model of outbreak behaviour has deep precedent in the extensive literature on mountain pine beetle (*Dendroctonus ponderosae*, Hopkins), where natural enemies are thought to play a diminished role in the release and collapse of outbreaks (Powell and Bentz 2014; Duncan et al. 2015). Second, at the other end of the spectrum, in other forest lepidopteran systems where cycling is more regular and better synchronized, occurring every 9–13 years, outbreaks only last for 1–3 years, and the role of specialist natural enemies in cycle induction is much clearer (Myers and Cory 2013). The consensus working hypothesis is that the SBW system is a hybrid system lying somewhere between these two extremes.

Preventability of budworm population irruption is also a practical matter of scale and economics (see section on Management - Early Intervention Strategy, AS 10 and 11). An irruption that is preventable at the level of the tree, stand or focal epicentre may not be preventable at the level of the landscape, because there may be economic, regulatory or logistical barriers (including lack of access, or inefficient access across large geographic areas) to upscaling an intervention program (i.e., jurisdictions or regions where the cost of, resistance to, or infeasibility of spraying insecticide across a wide area is prohibitively high). Prevention requires continuous early detection, and the challenging logistics and economics of early detection (a continuous annual program of monitoring using pheromone traps, and branch sampling for larvae) can, for some forest management operations, be a formidable barrier to pursuing the EIS. What is feasible for a small, highly timber-dependent province such as New Brunswick may be practically and economically infeasible for more diversified economies in larger provinces such as Quebec or Ontario, where outbreaks cover areas that are orders of magnitude larger. These issues are explored more fully in the comprehensive reviews by Johns et al. (2019) and Régnière et al. (2019a).

In summary, localized population irruptions at the level of the tree, stand or watershed may be indefinitely preventable. Larger-scale, region-wide outbreaks may be preventable for a certain amount of time (MacLean et al. 2019), but it remains to be seen whether the approach can be applied long-term. At the scale of areas the size of Ontario or Quebec, cycling behaviour may eclipse all locally irruptive behaviour, but evidence for this proposition, once thought to be strong, is gradually weakening with time.

Uncertainty

1. High uncertainty on the spatial scale across which this irruptive behaviour, and this preventability proposition, holds. It is likely that the feasibility of preventability is reduced as the area-wide population cycle rises in the surrounding regions. The level of uncertainty on the preventability of SBW population irruptions is low at the smallest spatial scales but moderate at the landscape scale.
2. We think that areas the size of New Brunswick, the Côte-Nord region of Quebec or Newfoundland might be amenable to outbreak prevention, but this is an area of moderately high uncertainty, where research needs to be targeted with high urgency. At the largest spatial scales—areas many times the size of New Brunswick- such as Ontario or Quebec—uncertainty drops back to moderate, as evidence in favour of preventability switches from strongly positive to insufficient to evaluate. The evidence here is considered insufficient because (a) our changing state of knowledge on the robustness of the cyclic component of outbreak behaviour has us rejecting a proposition we once found compelling, and (b) the barriers to intervention at this scale are hypothetical, involving historical experience with logistical and economic constraints.
3. The level of uncertainty on SBW population dynamics is low relative to the needs of most forest managers. However, the level of uncertainty may be considered higher when compared to the demands of some forest managers. For example, implementing an EIS requires more data and a clearer understanding of thresholds and their evolution in time and space than traditional pest management approaches.
4. Low uncertainty in the idea of an unstable irruption threshold above which budworm populations outbreak, and below which budworm populations decline to endemic. There is greater uncertainty on the estimate of this threshold and how it might vary in space, time, and ecological and climatic context.
5. Low uncertainty that as the barriers to large-scale intervention become numerous and pervasive with increasing size of the affected area, the ability to suppress populations becomes more challenging as irruption thresholds are breached.

Reducing uncertainty

1. More precisely quantify factors affecting SBW population ecology around the critical thresholds that determine when SBW populations may reach outbreak levels.
2. Characterize and quantify the behavioural mechanisms and impacts of moth dispersal on the spread and collapse of outbreaks, as mediated by climatic conditions and forest structure.
3. Characterize the factors affecting the transition from focal epicentres to large-scale outbreak using synthetic spatial modelling and validation. So much of our "understanding" comes from theoretical ecology

as opposed to field measurement. This is an urgent area of research if we are to predict budworm risks in the 10–15-year time horizon.

4. Develop remote sensing technology aimed at detection of low-level SBW defoliation.
5. Quantify the influence of stand structure, in particular foliage density (per ha), on SBW population dynamics.

AS 3: Long-distance dispersal of SBW moths is a major factor in outbreaks (By B.J. Cooke, J. Régnière, J.-N. Candau)

Evidence

Key points:

- SBW dispersal occurs at a wide range of spatial scales, from metres to hundreds of kilometres.
- Understanding of the environmental factors affecting long-range dispersal has improved considerably, with canopy observations of take-off flights and radar observations of atmospheric transport.
- Mechanistic models of moth dispersal indicate that much of the dispersal of SBW moths is wind-driven and therefore that wind direction and wind speed play a critical role in determining moth flight paths.
- There appear to be two major ways in which dispersal can contribute to cycling and spread. First, immigration events can trigger irruptive growth and spread. Whether this happens depends on the receiving population already being close to the irruption threshold. Second, immigrations between neighbouring populations can lead to a synchronization of cycling by reducing the delay between otherwise independently cycling populations.
- There are cases where the arrival of SBW moths from Quebec coincided with the establishment of novel populations and the initiation of focal epicentres. At the same time, it is important to underscore that not all immigration events result in expansion of the source outbreak, or irruption at the destination receiving immigrants. The propensity for dispersal-driven irruptions thus appears to depend on factors governing cycling behaviour at the destination location.

Details:

It has been known for decades that SBW dispersal occurs at a wide range of spatial scales, from metres to hundreds of kilometres, with early observations of long-distance flights of immigrating moths occurring as far back as records have been kept (reviewed by Blais 1953). During the previous outbreak cycle of the 1970s, understanding of the environmental factors affecting long-range dispersal improved considerably, with canopy observations of take-off flights and radar observations of atmospheric transport (Greenbank et al. 1980). Royama (1984) showed that the single largest source of variation in SBW generation survival at any given point is the ratio of all eggs to local moths, which is determined principally by the net gain or loss of female moths from immigration or emigration.

Because of the complexity and difficulty of making direct quantitative observations of moth dispersal, inferences about the role of adult dispersal in the population dynamics of budworm involves contrasting real-world observations and modelled understanding of all the other factors contributing to population dynamics. The ability to evaluate the long-range dispersal hypothesis is thus constrained by the quality of the models used in the contrast, and of the data available for model validation.

Early models of the role of SBW dispersal in population dynamics considered the likely influence of simple diffusive dispersal on cycle amplitude (Royama 1979) and synchronization (Régnière and Lysyk 1995). However, more recent mechanistic models of moth dispersal indicate that much (but not all) of the dispersal of SBW moths is wind-driven and therefore that wind direction and wind speed play a critical role in determining moth flight paths (Sturtevant et al. 2013; Régnière et al. 2019b, c; Garcia et al. 2022). This is consistent with Royama's (1984) inference about dispersal as a major source of randomness in population fluctuations at a given point. The more advanced mechanistic spatial population models suggest that dispersal interacts with population growth to produce a complex array of effects (summarized in Cooke et al. 2007; Sturtevant et al. 2015).

There are cases, such as the island of Newfoundland, where the arrival of SBW moths from Quebec coincided with the establishment of novel populations and the initiation of focal epicentres (Rhainds et al. 2019). At the same time, it is important to underscore that not all advective immigration events result in expansion of the source outbreak, or irruption at the destination receiving migrants (Sturtevant et al. 2013; Régnière, unpublished data). The propensity for dispersal-driven irruptions thus appears to depend on factors governing cycling behaviour at the destination location. James et al. (2015) showed that at the start of a SBW outbreak cycle, the meta-population has a patchy genetic structure that is relatively unmixed, but as the focal irruption grows into a large-scale outbreak cycle, population genetic structure becomes homogenized through dispersal. This is consistent with the proposition that dispersal is density-dependent, as first discussed by Royama (1979) and then demonstrated by Régnière and Nealis (2019a). Any process that is strongly density-dependent is going to influence population cycling and the spatial dynamics of outbreak spread.

Each year, new data accrues providing more detail on the major factors affecting flight propensity—in particular, female flight propensity (Rhainds 2015; Dargent et al. 2023), which remains the single biggest gap in improving our ability to model the effect of dispersal on producing both travelling waves of irruption and cycles that are synchronized across thousands of kilometres.

In summary, there appear to be two major ways in which dispersal can contribute to cycling and spread. First, immigration events can trigger irruptive growth and spread. Whether this happens depends on the receiving population already being close to the irruption threshold. Second, immigrations between neighbouring populations can lead to a synchronization of cycling by reducing the delay between otherwise independently cycling populations. We assert that it is not possible to understand the spatial pattern of SBW outbreaks without understanding and including the dispersal of gravid females with the many other factors affecting local irruptivity and cycling.

To the extent that weather is random from year to year, longer-term forecasts (where highly variable annual forecasts, aggregated, converge on prevailing wind fields) may be more robust. Working against this, however, is the opposite fact that population forecasting becomes decreasingly reliable with increasing time horizon (see section on Population dynamics and outbreaks, AS 1). This matters because spread is the consequence of the interaction between growth and dispersal. If forecasting accuracy decreases in one, while increasing in the other, then the net result is inescapable uncertainty across all prediction time horizons. Consequently, we conclude that there is generally high uncertainty on the ability to forecast dispersal-driven population phenomena.

1. High uncertainty in our ability to accurately simulate in real time (or historically backcast) long-range dispersal given weather data, but the uncertainty is decreasing quickly as new insights into environmental factors shaping dispersal emerge, and as availability of well-resolved weather data products rises.
2. High uncertainty in our ability to detect "hotspots" (locations where SBW density is near the irruption threshold) but modelling dispersal in near-real time during the flight season or soon after it could decrease uncertainty by pointing to areas where immigrant egg-carrying moths may have landed.
3. Long-range flights of immigrating moths are dependent on seasonal weather and weather during adult emergence and flight and are therefore impossible to forecast much in advance. Thus, there is high uncertainty on medium-range weather forecasts (e.g., one year in advance), and it is unlikely that this uncertainty can be reduced.
4. There is generally high uncertainty about the ability to forecast dispersal-driven population phenomena such as irruptions or travelling waves, which are the product of dispersal and nonlinear population growth.
5. The limited ability to accurately advance or forecast population behaviour and long-range immigrations in the 10–15-year time horizon means there is high uncertainty on any one forecast as to impacts during this time horizon. This means it will always be necessary to model impact scenarios and to use adaptive management whenever observations are deviating from one assumed scenario to another.
6. Low uncertainty that long-range dispersal of female moths is an important factor affecting both irruptive spread of outbreaks from epicentres and the synchronization of cycles.

Reducing uncertainty

1. Initiate ecophysiological research into factors governing take-off and capacity to sustain flight to improve forecasting the early part of the flight. Similar needs exist for wing folding and descent, followed by host-choice behaviour, particularly for night flight and when moths are forced down by convective storm fronts.
2. Improve dispersal models (including generalized kernels derived from ensembles of individual-based simulation model outputs) to include local/regional factors (i.e., topography, sea breeze effects), and move from a trajectory to a dispersion model (i.e., individual-based to concentration-based) for better spatial and temporal resolution.
3. Enhance understanding of differences between male and female moths in all aspects of flight behaviour.

4. Improve capacity for observational monitoring through a better integration of the various monitoring tools (i.e., traditional and automatic pheromone traps, light traps and radar) to test predictive flight models.

5. Develop enhanced methods of distinguishing resident populations from immigrants (e.g., stable isotope screening).

Consequences of Climate Change

AS 4: Climate change is altering the distribution, scale, and intensity of SBW outbreaks in the forests of eastern Canada

(By A.D. Roe, D.S. Pureswaran, J.J. Bowden, J. Régnière, E.R.D. Moise, V. Martel)

Evidence

Key points:

- Climate change will alter thermal regimes, variability, and the frequency and duration of extreme weather events in the range of SBW.
- Spruce budworm is sensitive to climatic conditions, and these conditions determine SBW's range limits.
- Synchrony with host plants is temperature-dependent and is crucial for SBW survival.
- Spruce budworm supports a diverse community of natural enemies which, are also sensitive to climatic conditions.
- SBW outbreaks have shown recent northern expansion into naïve black spruce forests north of where outbreaks occurred in the past.

Details:

Spruce budworm is sensitive to climatic conditions, with temperature, precipitation and wind all playing a role in the frequency, duration, severity and synchrony of population outbreaks (Régnière et al. 2012; Gray 2008, 2013; Pureswaran et al. 2016; Williams and Liebhold 1997). Climate change is already altering and will further alter thermal regimes, average conditions and climatic variance, including the frequency and duration of extreme events. Mean annual temperature in northern regions of eastern boreal forests could increase between 3°C and 6°C in the next 20–50 years (Ouranos 2015), thus exposing SBW populations, and the boreal forest communities, to new climatic conditions. These changes influence outbreak dynamics in irruptive insects such as SBW (Ekholm et al. 2020; Fleming and Volney 1995; Lehmann et al. 2020; van Asch and Visser 2007), but will be difficult to predict (Pureswaran et al. 2018).

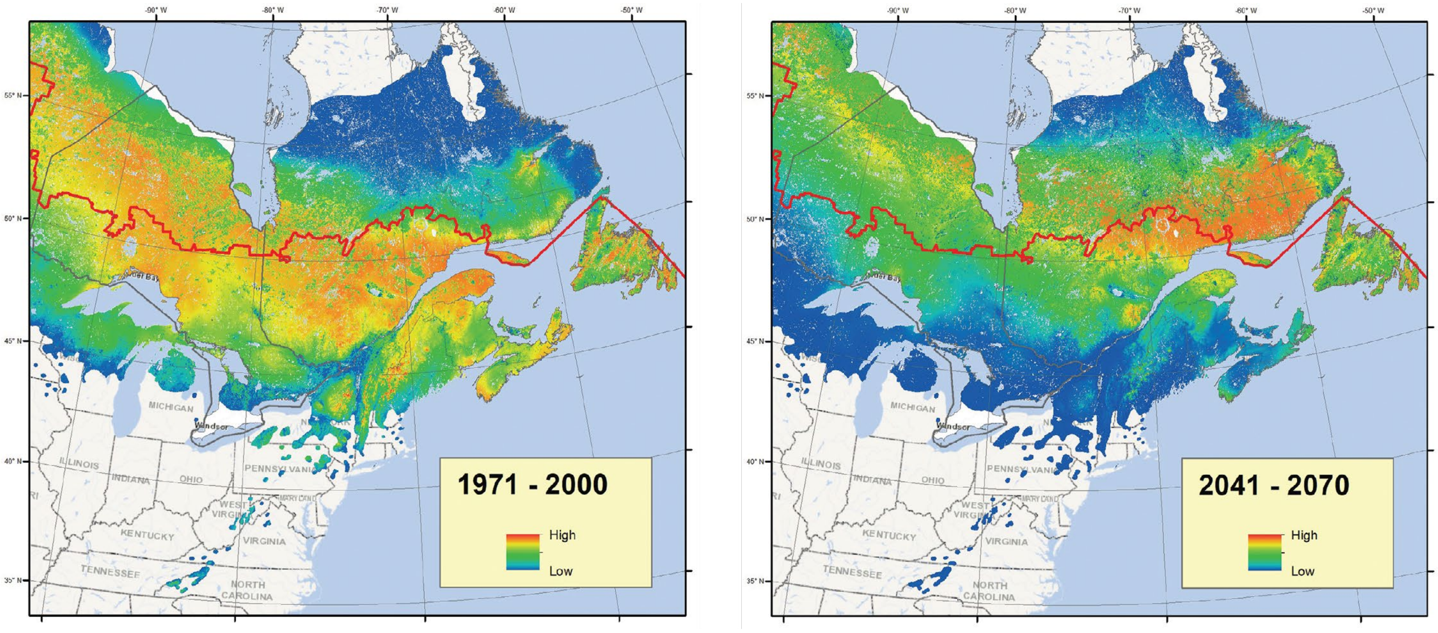

Figure 2. Maps of spruce budworm population growth rate index based on % host plant forest canopy cover under A. current (1971–2000) and B. future (2041–2070) climate conditions (modified from Régnière et al. 2012). Northern margin of commercial forestry and approximate boundary of forest health monitoring shown in red (from McKenney et al. 2016).

The SBW life cycle is optimized to a particular thermal range (Régnière and Nealis 2007, 2008), and climate change is expected to move that optimum northward (Fig. 2; Régnière et al. 2012; Candau and Fleming 2011; Pureswaran et al. 2015). Both the northern and southern limits of SBW are determined primarily by temperature. In the north, development time (Régnière et al. 2012) and cold tolerance (Gray 2008; Marshall and Sinclair 2015; Butterson et al. 2021) limit SBW populations. Length of the growing season determines whether SBW can complete its life cycle and enter its cold-tolerant stage before the onset of winter. Longer seasons would allow SBW to establish in habitats at higher latitudes and elevations (Régnière et al. 2012; Régnière and Nealis 2019b). Warmer winters are expected to reduce overwintering mortality in northern regions and improve recruitment success in the spring (Gray 2008). These distributional changes are predicted to occur during the next 50 years (Régnière et al. 2012; Gray 2008); however, current SBW outbreaks in Quebec are already occurring farther north than previously recorded (Pureswaran et al. 2015) and are causing defoliation and mortality in black spruce forests not previously exposed to SBW disturbance (Bognounou et al. 2017; Jacques Duval pers. comm.). Therefore, it is imperative that we improve our understanding of the ecophysiological response of SBW to climatic changes, as well as SBW-host tree interactions, to prepare for new future dynamics.

Spruce budworm larvae survive winter in a state of suspended development called diapause (Han and Bauce 1993; Marshall and Roe 2021). They enter this state in late summer and do not resume development until spring temperatures cue their emergence (Régnière 1990). Induction and termination of diapause are critical life history stages for long-term persistence of insect populations (Marshall and Roe 2021) and are often influenced by winter temperatures (Lehmann et al. 2017). Diapausing SBW are well adapted to survive low winter temperatures by converting energy reserves to cryoprotectants. These same reserves, however, are critical to resuming larval development in the spring, and larvae experience high pre-emergence mortality under elevated spring temperatures, likely due to energy depletion (Han and Bauce 1998; Nealis and Régnière 2016). Under historic climatic conditions, overwintering mortality did not appear to vary year to year in northeastern Ontario or central Quebec (Régnière 1990; Régnière and Duval 1998) nor contribute significantly to population dynamics (Régnière and Nealis 2008). New climatic conditions, however, may alter this dynamic. Climate change is more pronounced in winter, particularly at high latitudes (Price et al. 2013; Marshall et al. 2020), so SBW populations will need to cope with elevated temperatures and increased variability (e.g., Liu and Zhang 2020; MacQuarrie et al. 2019). Although warmer winters may improve overwintering survival at the northern edge of the range (Régnière et al. 2012), the impact on SBW populations will be spatially and temporally variable. For example, we expect to see increased mortality and reduced recruitment in the spring along the southern margin of SBW's range as higher temperatures during winter and increased variability cause greater energy depletion during diapause (Régnière et al. 2012; Nealis and Régnière 2016; Roe et al. 2024), although some models predict no change in the southern limit of the outbreak range (Candau and Fleming 2011; Gray 2013). These populations may also be more susceptible to extreme cold events due to depleted energy reserves, compounding the climate's effects on survival. Temperature extremes and variability may interact to produce lethal or sub-lethal effects that decrease overall population performance (Marshall and Sinclair 2015; Marshall et al. 2020). Ultimately, it is uncertain how resilient overwintering SBW are to these climatic changes and whether they will affect the scale and intensity of SBW outbreaks at the northern and southern edges of the current distribution (Candau and Fleming 2008; Diamond and Yilmaz 2018; Gray 2008; Régnière et al. 2012).

During the last 70 years, temperatures significantly increased within the SBW range with the greatest increase occurring during summer (+0.21°C per decade), followed by spring (+0.17°C per decade) and fall (+0.12°C per decade) (Boulanger et al. 2024). As a result, peak abundances of L2 and L4 occurred up to more than 20 days earlier in 2022 when compared with 1951, notably in the western part of its range. These changes resulted in a notable northward expansion of suitable climate conditions for SBW during this period (Boulanger et al. 2024), with higher reproduction rates in the northern regions. In contrast, the southern areas of its range are seeing increased simulated winter mortality due to rising temperatures (Roe et al. 2024). Consequently, suitable growth rates have moved northward by an average of more than 68 kilometres, with this shift exceeding 200 kilometres in the easternmost parts of its range (Boulanger et al. 2024). Such recent spatial changes in SBW climate suitability could explain the rather northward location of the current outbreak in eastern Canada. It also means that climate change impacts on SBW dynamics are not for tomorrow: they are already here.

The northward shift of SBW populations exposes black spruce forests to increased risk of defoliation and mortality (Pureswaran et al. 2015; Bognounou et al. 2017). In the past, initial outbreak foci began in southern forests and moved northward towards the end of the outbreak, with relatively low impact on black spruce forests. However, the current outbreak in Quebec began further north, where black spruce is dominant or co-dominant with balsam fir (Bouchard and Auger 2014; Pureswaran et al. 2015). Temperature tightly controls the phenology of SBW and its host trees (Portalier et al. 2022). The timing of emergence from diapause is cued by spring temperatures and occurs prior to budburst in its host plants. Synchrony with each host's phenological window strongly affects fitness in SBW populations (Nealis and Régnière 2004; Régnière and Nealis 2018; Portalier et al. 2024). Black spruce typically experiences budburst later than other hosts, leading to high mortality in early instar larvae, and this usually protects black spruce from high defoliation through the phenological mismatch (Nealis and Régnière 2004). Temperature increases associated with climate change may alter the timing of these events, shifting the phenological window and creating greater synchrony between SBW and black spruce (Pureswaran et al. 2019). When phenologically synchronized, SBW shows similar survival on black spruce as balsam fir, so it is expected that black spruce would support higher populations of SBW under these circumstances (Fuentealba et al. 2017, 2018; Bellemin-Noël et al. 2021). The ecological impact of high SBW defoliation on black spruce-dominated ecosystems under new climatic conditions is not known. Forest succession following black spruce mortality would depend on the proportion of fir and deciduous species, fire recurrence, and regeneration patterns that could transform the forest to more productive mixedwood or less productive ericaceous shrublands (Pureswaran et al. 2015). Habitat transformation could have cascading effects on higher trophic levels, including boreal caribou (Labadie et al. 2021).

The SBW system supports a diverse community of natural enemies (Eveleigh et al. 2007). The impact of climate change on the interactions between these community members is multifaceted and ecologically complex (Fleming and Volney 1995; Volney and Fleming 2000, 2007). Temperature affects the developmental time and thus the phenology of all insects, including parasitoids. Multivoltine parasitoids have more than one generation per year and must be synchronized with more than one host species per year; climate change will alter the phenological timing of these interactions, resulting in unforeseen, cascading effects through the system. However, models predict that, like SBW, climate change will cause a northward shift of at least two multivoltine species of SBW larval parasitoids, *Tranosema rostrale* and *Meteorus trachynotus* (Régnière et al. 2020, 2021), allowing them to track distributional shifts in their host. Conversely, the southern range of these parasitoids is projected to become less suitable. Although all members of the SBW community will be affected by climate change, these changes will not be consistent in magnitude or direction among species. The community-level impacts of these perturbations could result in new or altered tri-trophic interactions, especially as SBW expands into northern forests. The ability of parasitoids and other natural enemies to control SBW populations and alter their population dynamics in northern forests requires further study (see also section on Population dynamics and outbreaks, AS 1 and 2).

Temperature also affects parasitoids indirectly by affecting their hosts' immune response. *Tranosema rostrale* has higher mortality caused by the insect's immune response when reared at high temperature (Seehausen et al. 2017). The cause of this mortality was identified as a combination of a reduced expression of polydnavirus genes injected by the mother at oviposition and an enhanced expression of immunity genes in SBW (Seehausen et al. 2018). The host plant is also known to affect the immune response of herbivorous insects (Vogelweith et al. 2011). Consequently, the impacts of temperature on parasitoid performance could depend on the host plant, but again this interaction is not well understood.

In the current outbreak in Quebec, we are seeing a northward range expansion of SBW populations. During the coming decades, climate change and its impact on SBW populations will increase, along with the uncertainty of the direct, indirect, and cascading effects on the whole SBW system. Critical baseline data are needed on ecophysiological responses to climate, within SBW and the relevant biological community, to understand how the system will respond to future environmental conditions, as well as the complex ecological mechanisms driving this response. Standardized, empirical data obtained via long-term monitoring is critical for understanding the impact of climatic fluctuations and changes on insect populations due to the complexity of effects across different temporal and spatial scales (Halsch et al. 2021). Current monitoring efforts are focused on commercial forests in the southern regions of the boreal forest, regions currently at high risk of SBW outbreaks. To prepare for a future under new climatic conditions, however, we need baseline data to help predict the magnitude of change at range margins and to reduce uncertainty in population dynamics models.

Uncertainty

1. High uncertainty on the vulnerability of black spruce forests to SBW disturbance under climate change and the resulting impacts, with moderate to low uncertainty of the northward movement of a more favourable climate regime for SBW.
2. High uncertainty about how climatic changes will influence SBW population dynamics via natural enemy control and other mortality factors.
3. High uncertainty regarding the impact of climate change on the synchrony between parasitoids, their hosts (including SBW), and the host plant.
4. Moderate uncertainty in the thermal limits of SBW during diapause (fall, winter, spring), particularly with increased frequency of extreme weather events.
5. Moderate uncertainty regarding the impact of increasing temperatures or extreme events on SBW populations at the southern margin of the species.
6. Moderate uncertainty in the direct impact of climate change on host plants of SBW.
7. Low uncertainty that SBW populations will expand northward into black spruce dominated forests.
8. Low uncertainty that climate change will increase the probability and frequency of SBW outbreaks in the north and decrease them in the south.
9. Low uncertainty in the thermal requirements for SBW growth and development.
10. Low uncertainty that higher temperature increases the SBW immune response, and consequently mortality rates in some parasitoids.

Reducing uncertainty

1. Quantify the impact of temperature (means, variability) on SBW's survival, physiology, and life history, particularly at range margins under future climatic conditions.
2. Characterize SBW disturbance in black spruce forests under future climatic conditions and quantify the impacts.
3. Improve monitoring in northern areas, beyond managed forests, to measure SBW population changes and impacts on host tree growth, survival, and forest succession.
4. Quantify the impact of temperature on the developmental time of different SBW parasitoid species and their main alternate hosts, and development of prediction models for the performance and distribution of SBW parasitoids under different climate change scenarios.
5. Quantify the interaction between temperature and host trees on the SBW immune response and parasitoid survival.
6. Improve the ability to forecast budburst phenology and SBW L2 emergence to predict changes in phenological synchrony and defoliation risk.

Interaction with fire

AS 5: SBW outbreaks increase risk of wildland fire (By L.M. Johnston, Y. Boulanger)

Evidence

Key points:

- Fire and SBW outbreaks are both common disturbances in much of eastern Canada, and they are linked both directly and indirectly with each other.
- Damage from SBW can result in a buildup of fuel for fire to burn, and the changes to the physical fuel arrangement have the potential to dramatically increase fire risk at the stand level and, under certain conditions, can produce extreme fire behaviour.
- There appears to be a temporal lag of effects of SBW on fuel availability; a peak occurs several years after the end of the defoliation event, creating a "window of opportunity" for increased fire risk.
- The relationship is linked in many ways and can be modified by weather/climate, stand composition, fire regimes, fire suppression, and fuel or stand dynamics, changing seasonally or gradually with time.
- Better understanding of the SBW/fire relationship will be even more crucial in the future, with the dynamic changes that SBW, its hosts, and fire are facing due to climate change.

Details:

There is a large amount of work done on addressing the question of whether SBW outbreaks can increase the risk of wildland fire in eastern Canada. Damage from SBW can result in a buildup of fuel for fire to burn (Brassard and Chen 2006; Watt et al. 2018; Watt et al. 2020; Sato et al. 2023). In addition to the increase of dry, available fuel, the changes to the physical fuel arrangement (in particular, vertical fuel continuity) have the potential to dramatically increase fire risk at the stand level (Stocks 1987). With these fuel changes, both primary components of fire risk (likelihood and impacts) may be affected. There appears to be a temporal lag of the effects of SBW on fuel availability; a peak occurs several years after the end of the defoliation event, creating a "window of opportunity" for increased fire risk (Stocks 1987; Fleming et al. 2002; James et al. 2017; Watt et al. 2020). These dynamics have been shown to vary spatially; Fleming et al. 2002 discovered a longitudinal effect on the time between outbreak and fire occurrence. In fact, in some areas with limited fire activity (i.e., wetter climates), the climate-driven rate of decomposition can outpace the fuel buildup, thus eliminating the period of increased fire risk (Péch 1993).

Fire behaviour[4] in stands affected by SBW was studied empirically in a set of experimental burns in Ontario during 1976 to 1982, as described in Stocks (1987). These fires were ignited in mixedwood plots consisting primarily of dead balsam fir and partially defoliated black spruce (Fig. 3a and 3b). Ultimately, the fuel consumption and fire behaviour documented in these experiments formed the basis of two "fuel types" in the Canadian Forest Fire Danger Rating System, which provides quantitative estimates of fire behaviour for use in operational fire management (Forestry Canada Fire Danger Group 1992; Wotton et al. 2009). The "M-3 dead balsam fir mixedwood – leafless" fuel type is derived from burns done in spring before deciduous leaf-out decreases the fuel hazard. These burns exhibited extreme fire behaviour: ground ignition immediately transitioning to intense crown fire, very fast rates of spread, burning material being ejected and starting spot fires far outside of the fire area, and very high fuel consumption (Fig. 3c and 3d). Summer burns done after deciduous leaf-out ("M-4 dead balsam fir mixedwood – green") showed more limited fire behaviour; plots that were affected by SBW recently (i.e., less than 4 years) could not sustain fire spread, although after 4–5 years of fuel accumulation, one plot was observed to sustain fire spread and burned with very high intensity. Along with the other spring burns, even this summer burn exhibited fire intensity level above the threshold of possible control (Stocks 1987; Hirsch and Martell 1996). This very extreme fire behaviour and desire to re-establish affected stands prompted McRae (1986) to develop methods to bulldoze the affected stands and burn as a slash fuel. In SBW-damaged stands, there is the potential for much more extreme fire behaviour than what has been observed outside of eastern Canada with both mountain pine beetle (Perrakis et al. 2014) and western spruce budworm (Hummel and Agee 2003); this is likely due to the vertical continuity of fuels (i.e., "ladder fuels") that results from SBW damage (Perrakis et al. 2014; Watt et al. 2018).

Fire behaviour influences wildland fire risk, with extreme fire behaviour increasing both the likelihood and the

[4] Fire behaviour is how "…fuel ignites, flame develops, and fire spreads and exhibits other related phenomena as determined by the interaction of fuels, weather, and topography" (Canadian Interagency Forest Fire Centre 2021).

impacts of fire. Fire risk, however, is influenced by many factors, and there are a variety of metrics that can be investigated to characterize risk (Johnston et al. 2020). The effects of insect outbreaks on variables such as fire ignition, area burned, number of fires, fire size, and fire severity have been investigated, but with this wide range of variables (along with differing spatial and temporal scales), these studies have not led to consensus on the topic (James et al. 2017; Watt et al. 2020). Fleming et al. (2002) investigated the proportion of area burned in areas of Ontario that had seen SBW damage between 1941 and 1996 (as described by Candau et al. 1998). They found that areas with damage occurring with moderate frequency (9–11 years in total) had experienced a higher proportion of area burned by large fires; they suggest aggressive fire suppression is responsible for limiting fire activity in low frequency outbreak areas, which are typically closer to more urban areas. Further work found that there is a high probability for SBW and fire to interact in a large part of Ontario, with that interaction dependent on climate moisture and frequency of defoliation (Candau et al. 2018). In such areas, it was found that SBW damage promoted subsequent large fires. This area of interaction appears to be spatially bound by climate and stand composition, but also possibly by fire activity (as modified by climate or fire suppression actions). The effects of SBW outbreaks on fire ignition by lightning have also been investigated; James et al. (2017) studied two budworm-killed areas of Ontario and found that ignition probability could either increase or decrease, with dependence on temporal scale, ecoregion, and whether it was spring or summer. Notably, there was a clear temporal effect, with recent defoliation decreasing the ignition probability, but then after several years of fuel accumulation (as also discussed in Stocks 1987 and Fleming et al. 2002), the ignition probability increased (James et al. 2017).

Figure 3. Images of SBW-damaged stands used in experimental burning 1976–1982 at the Aubinadong River, Ontario; (a) and (b) show the mixedwood stands with dead balsam fir and spruce in the understory, and the overmature mixedwood overstory, (c) and (d) depict typical crown fire behaviour exhibited in the "M-3 dead balsam fir mixedwood – leafless" fuel type (all photos from Great Lakes Forestry Centre fire research slide library).

Fire and SBW outbreaks are both common disturbances in much of eastern Canada (Fig. 4), and they are linked both directly and indirectly with each other, but also with other disturbances such as storm damage, drought, and other damaging insects or pathogens (Bouchard et al. 2006; Boulanger et al. 2013; Gray 2013; Buma 2015; Navarro et al. 2018). Fire can also affect vulnerability to SBW outbreaks, notably by impeding the short- and medium-term regeneration of the most vulnerable hosts, i.e., balsam fir (Bergeron and Leduc 1998; Bouchard and Pothier 2008), with heterogeneous interactions across the landscape mosaic of stand ages, disturbance histories, and forest conditions (Brassard and Chen 2006). Further work is needed to understand these interactions and the importance of how the relationship between SBW and fire can be mediated by a variety of factors and local conditions. Given the overarching importance of climate/weather on these relationships and the large range of climate-dependent fuel availability (e.g., compare the results of Péch 1993 and Stocks 1987), it is likely that the relationship between fire and SBW will become more complex and dynamic under the effects of climate change (see section on Consequences of climate change, AS 4).

Remote sensing tools and a continued effort to characterize the impacts of SBW and of fire will help to improve tools that integrate these two disturbances (e.g., fuel type mapping; Guindon et al. 2014; Simpson 2014). With the many challenges faced in fire management and forestry, reducing uncertainty, and improving the understanding of the risk to forest values and to communities is crucial for successful management and planning (Boulanger et al. 2016; Johnston et al. 2020). Better understanding will be even more crucial in the future, with the dynamic changes that SBW (Fleming and Candau 1998; Candau and Fleming 2011; Régnière et al. 2012; Boulanger et al. 2016; Boucher et al. 2018), its hosts (Boulanger and Puidevall 2021), and fire (Wang et al. 2017; Wotton et al. 2017) are facing due to climate change.

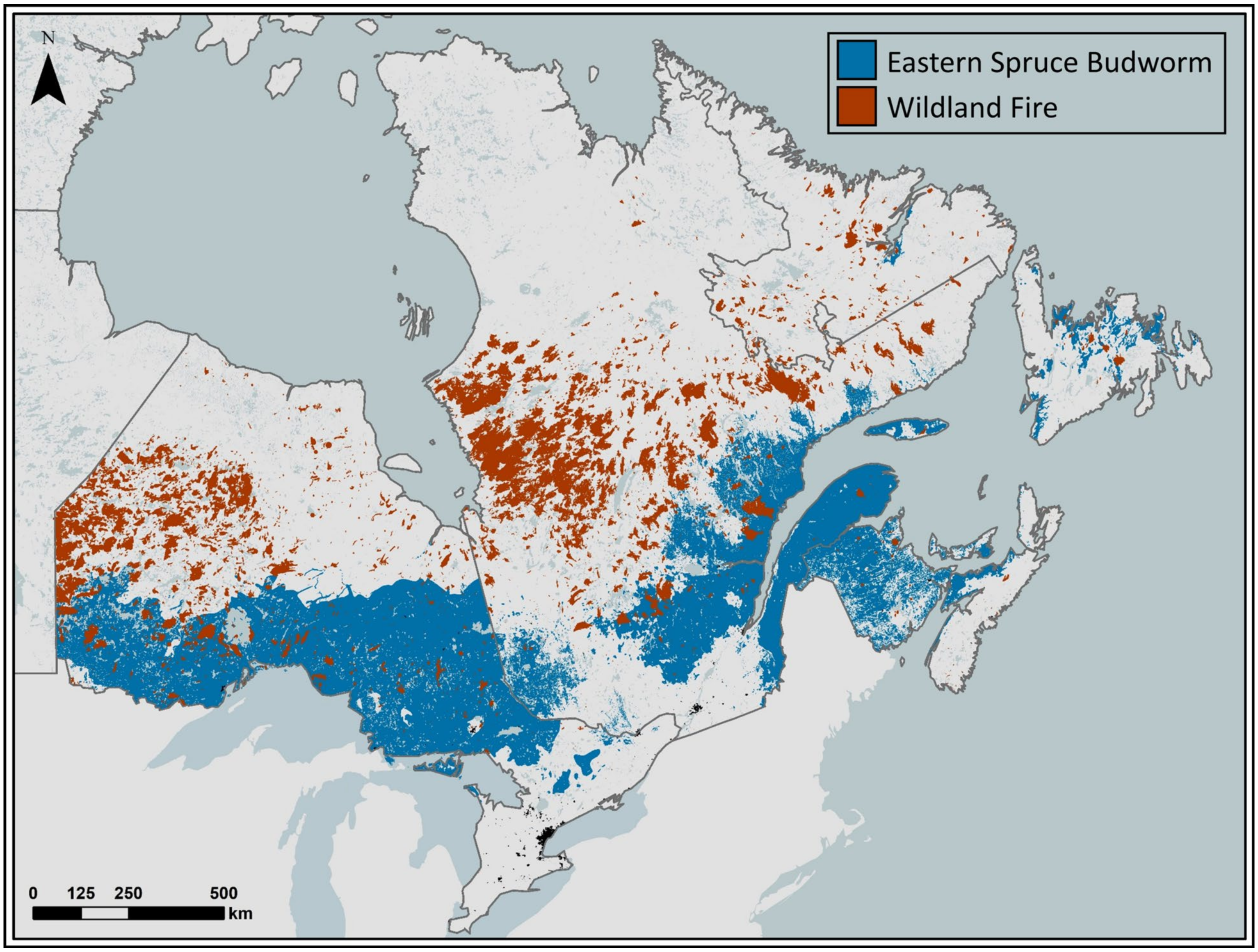

Figure 4. Map of area burned by wildland fire (data from the Canadian National Fire Database, Natural Resources Canada) and the extent of moderate and severe SBW outbreaks for 1980–2020 (Pest Strategy Information System, Natural Resources Canada, Atlantic Forestry Centre). Note that some provinces are missing disturbance data for multiple years, and area burned may exclude many small fires (e.g., some provinces only report >200 ha fires).

Uncertainty

1. High uncertainty of how wildland fire risk is affected at a landscape scale under a variety of outbreak severities and frequencies, and the impact of spatial and temporal scales on the controls of these relationships.
2. Moderate uncertainty that stands affected by SBW can have a higher likelihood of wildland fire.
3. Low uncertainty that SBW can increase wildland fire behaviour (i.e., extreme spotting, spread, intensity) and exceed the limits of fire suppression.
4. Low uncertainty that SBW can modify the fuels available to burn in a wildland fire.

Reducing uncertainty

1. Improve characterization of SBW fuel types and integrate these fuel types into operational tools and fuel maps through empirical research and monitoring.
2. Assess how wildland fire metrics are affected by SBW damage, and the implications of the impacts on fire risk.
3. Investigate cumulative and cross-scale interactions between SBW and fire, and consideration of how climate and fire management modify these interactions through time.
4. Develop hazard reduction options (e.g., fuel breaks, fuel modification), including assessment of the impacts of salvage logging in SBW-affected stands on fire risk.

Impacts of outbreaks

AS 6: SBW outbreaks can result in negative impacts on market and non-market forest values (By D.W. McKenney, E.S. Hope, V. Lantz)

Evidence

Key points:

- SBW outbreaks reduce the available supply of merchantable spruce-fir fibre in affected forests, causing downstream (e.g., shift in supply of spruce products) and upstream (e.g., stand harvest implications) economic impacts.
- From an economic perspective, SBW should be managed when the benefit of management efforts exceeds associated costs.
- Economic considerations should include pest population dynamics, market values, feasible treatment strategies, harvest scheduling, the prioritization of optimal stands, and the associated ecosystem benefits of affected forest areas.
- Economic considerations can be expanded to include non-market values, including, for example, ecosystem services, carbon sequestration, and habitat provision. These non-market values can provide additional support for SBW management. Quantifying non-market values is an ongoing process, and subject to significant uncertainty for both validity and reliability.

Details:

Forest disturbances are a natural part of Canada's forest ecosystems; annual volume losses to insects have been estimated as roughly half the national annual harvest level (Hall and Moody 1994; Mushakhian et al. 2020), highlighting the impact forest pests have on fibre supply. Spruce-fir forests comprise a large proportion of the fibre for Canada's wood products industry (Natural Resources Canada 2020), hence, defoliation and eventual mortality caused by SBW outbreaks may lead to significant losses in fibre supply, affecting mills and non-timber, environmental, and social values associated with these forests (MacLean 1985; Chang et al. 2012b). Losses vary with the proportions of various host species in affected stands (differences in both susceptibility and vulnerability), and impacts on individual mills will depend in part on the particular host species utilized and the products produced. Repeated defoliation of needles on annual shoots reduces growth and often leads to mortality of host trees after several years of consecutive defoliation (Blais 1981; MacLean and Ostaff 1989). Balsam fir mortality of 75–100% usually begins after 5 years of severe defoliation, and white spruce mortality of 10–25% after 6–7 years of defoliation (Belyea 1952; Blais 1958; Elliott 1960; Batzer 1973; Howse et al. 1980; Blais 1981). The impacts of SBW outbreaks on fibre supply are arguably the most relevant from an economic context, although future efforts to value the carbon sequestered in SBW-affected forests may identify larger economic impacts (see section on Impact of outbreaks - Carbon sequestration, AS 7 D, for a discussion on the current state of carbon quantification from insect outbreaks).

Multiple methods and programs (aerial application of insecticides, salvage logging, harvest re-scheduling, EIS, etc.) exist to mitigate the losses associated with SBW. From an economic perspective, the main question is whether the costs of management efforts are justified such that the benefits of management are greater than the costs. The answer should be based on knowledge of pest population dynamics, stand dynamics, impacts on resource values, and the suite of available treatment strategies (Fox et al. 1997). Assessing the economic impact of SBW on available fibre supply is important at multiple scales given that outbreaks can grow from the scale of individual stands to extensive landscapes. Conceptualizing the economic impacts of SBW in this way sets the stage for investments in stand-to-forest management interventions and focuses on the idea of thresholds and optimal investment levels. Approaching SBW management via economic thresholds compares the presumed benefits of SBW management (avoided market and non-market losses) with the cost of implementing any particular management method/program (Fox et al. 1997).

SBW-related literature contains several assessments of the economic impacts and cost-benefit analyses of the insect and its management. The focus is often on the harvest queue problem—what stands to harvest when (e.g., MacLean et al. 2001, 2002; Mushakhian et al. 2020). Published research on optimal management suggests the need for prioritizing high-value, vulnerable timber stands, adequate planning, and sufficient SBW monitoring (e.g., Hennigar et al. 2013). Indeed, without SBW management, harvest volumes in New Brunswick were expected to decline 13 to 27%, translating into stumpage losses between $178 and $273 million, depending on the severity of the outbreak (Liu et al. 2019a). Expanding the analysis beyond the forest sector, Liu et al. (2019a) estimate $24.6

to $35.3 billion in domestic economic losses, highlighting the possible magnitude of the market value impact of SBW outbreaks without mitigative efforts. Interventions may reduce the size of SBW outbreaks and downstream economic effects by supporting relatively consistent harvest volumes in the short term, and possibly in the long term if the EIS is successful (Liu et al. 2019a). However, traditional programs (e.g., applying insecticides and replanning harvests to pre-empt the arrival of SBW) will eventually realize a decline in fibre supply in the long term (Hennigar et al. 2007; Slaney et al. 2010; Chang et al. 2012a). Hennigar et al. (2013) concluded that despite SBW management and loss mitigation measures, harvest levels could decrease by upwards of 10% for a 25-year period following a SBW outbreak.

Regardless of the scale of the outbreak, the available supply of merchantable spruce-fir fibre in affected forests will decrease during the long term as SBW defoliates and eventually kills timber stock (MacLean et al. 2002; Hennigar et al. 2007, 2013). The interdependency of fibre losses between stands is important, as stands partially affected by SBW may no longer contain sufficient volume to satisfy economic harvest calculations (i.e., harvesting a reduced volume is no longer profitable) (MacLean et al. 2002). Researchers have also suggested impacts on other forest-related sectors including recreation and tourism, which can affect the cost-benefit calculus (Chang et al. 2012b), but more so on carbon sequestration (Slaney et al. 2009). SBW management measures will be of greater interest with the emergence and use of carbon markets (see section on Impacts of outbreaks - Carbon sequestration, AS 7 D). Impacts of SBW on non-market ecosystem services such as wildlife habitat or water quantity and quality are difficult to quantify in both biophysical and human value terms. However, there is a growing literature in this area that seeks to quantify public willingness to pay for programs that mitigate the negative effects of SBW and thus preserve SBW-affected ecosystem services (Chang et al. 2011; Chang et al. 2012b). Chang et al. (2011) used a contingent valuation survey to estimate that, in New Brunswick, households would be willing to pay between $14.3 and $20.8 million annually for SBW control measures. There are no simple unequivocal indicators that provide normative economic (i.e., prescriptive) guidance involving trade-offs between various ecosystem services. Much depends on the demand for and value of standing timber, which, of course, varies spatially and temporally. Ultimately, the intensity, magnitude and spatial-temporal progression of outbreaks on stand and forest-wide mortality and/or growth will affect stakeholder viewpoints on management efforts and acceptable trade-offs (Chang et al. 2009).

Uncertainty

1. Moderate uncertainty on the interplay (trade-offs) between timber supply, carbon and SBW and optimal management efforts at medium to large spatial scales.
2. Moderate uncertainty related to the impact of SBW on trade-offs and optimal response regarding non-market values and severe SBW outbreaks (e.g., carbon, habitat supply and quality, water, fire risk).
3. Low uncertainty related to the general negative impact SBW will have on fibre supply during the long term in eastern spruce-fir stands, although the spatial patterns and specific forest management unit decisions are unknown until outbreaks occur.

Reducing uncertainty

1. Optimize timber management measures in regions with excess timber supply to better understand the required levels and locations of SBW management.
2. Improve knowledge on attitudes, preferences, willingness to pay, and costs of SBW management alternatives. This should help identify and justify SBW management programs and engage the public, directly affected stakeholders, and Indigenous peoples on options, opportunities, and trade-offs.
3. Better quantify biophysical measures/metrics and their non-market values to help with trade-off analyses in SBW management within timber supply and other forest management planning tools.
4. Better understand the dynamic (through time) nature and spatial patterns in trade-offs between market and non-market values with alternative (including targeted stand-level) SBW management efforts.
5. Incorporate near- to medium-term climate change scenarios in the trade-off analyses identified above for both market and non-market value impacts.

AS 7: Ecosystem integrity is generally resilient to natural SBW outbreaks but will alter ecosystem properties at shorter temporal and spatial scales (By L.A. Venier, M. Stastny, C.B. Edge, and J.J. Bowden)

Evidence

Key points:

- SBW outbreaks are an important natural disturbance across large areas in Canada, and region-wide outbreak suppression is expected to remove or reduce the natural, periodic imprint of SBW outbreaks not only on the age structure, composition, and regeneration of individual stands, but also on forest heterogeneity on the landscape, with poorly understood consequences on forest ecosystems.
- Anthropogenic change (e.g., climate change, harvest, introduction of non-native species, and post-SBW salvage logging) can interact with natural disturbances like SBW to move a system outside of its natural range of variation, with significant effects on biodiversity, nutrient cycling, and carbon sequestration.
- The pulse of SBW larvae during outbreaks supports the entire bird community, and the pulse of deadwood resulting from tree mortality supports saproxylic beetles that contribute to decomposition and to nutrient cycling, and that act as a key food source for birds and mammals. Salvage logging following SBW reduces the quantity of fresh deadwood, which in turn alters the saproxylic insect community and reduces foraging and nesting opportunities for bark gleaners and cavity nesters.
- Through a suite of changes associated with SBW defoliation (i.e., litterfall, canopy openness, tree growth and mortality, coarse woody debris, as well as stand dynamics), outbreaks have significant consequences for biogeochemical processes in affected forests on various timescales.
- A decrease in net ecosystem productivity (NEP) may convert boreal forests from C sinks into C sources after SBW outbreaks. However, some models have shown a rapid recovery trend in annual NEP if defoliation was reduced, suggesting that affected forests may return to acting as a C sink faster if defoliation intensity is not severe and long-lasting.

Details:

Ecosystem integrity

SBW outbreaks are an important natural disturbance across large areas of mixedwood and spruce-fir forests in boreal and hemiboreal Canada (Navarro et al. 2018). Within the historical range of these outbreaks, forest ecosystems are the products, in part, of the irruptive dynamics of SBW (Brandt et al. 2013), including biodiversity, regeneration and productivity, biogeochemical cycling, and landscape configuration of forests. By shaping and maintaining the forests, this natural disturbance is intrinsic to the definition of their ecosystem integrity, rather than affecting it. The most widely accepted forest ecosystem management paradigm aims to maintain ecological processes by emulating natural disturbances (Niemelä 1999; Spence 2001; Burton et al. 2006; Gauthier et al. 2009). For this reason, unmitigated SBW outbreaks do not necessarily pose a risk to the proper functioning of these forests and the ecosystem services that they provide if the outbreaks occur within the natural range of spatio-temporal dynamics and severity.

Anthropogenic change can interact with natural disturbance to move a system outside of its natural range of variation (Landres et al. 1999; Keane et al. 2009). First, climate change is expected to result in more northerly outbreaks of SBW (Figure 2; Régnière et al. 2012; Pureswaran et al. 2015). Defoliation is now occurring farther north in Quebec than previously observed (Bognounou et al. 2017), just as other irruptive boreal defoliators are shifting their outbreaks northward (e.g., Jepsen et al. 2008). Conversely, the southern regions may become climatically less suitable for sustaining SBW populations necessary to produce outbreaks (see section on Consequences of climate change, AS 4). Second, historical and present-day forest management has significantly altered forest composition and structure (Noseworthy and Beckley 2020) and host tree distribution. Harvest generally favours fir, whereas SBW outbreaks favour spruce on the landscape (Kneeshaw et al. 2022, Belle-Isle and Kneeshaw 2007). Landscape-level changes in forest attributes may in part explain the increase in the intensity of SBW outbreaks during the last cycle (i.e., the "silvicultural hypothesis"; Robert et al. 2012, 2018; MacLean 2015; Navarro et al. 2018; Cooke et al. 2024). Third, introductions of non-native species may interact with SBW outbreaks. For instance, post-outbreak sapling foraging by moose (*Alces alces*), a non-native species on the island of Newfoundland, can inhibit recruitment of balsam fir, causing long-term impacts on stand regeneration and composition (Gosse et al. 2011). Forest invasive alien species that include tree-feeding insects (e.g., Lovett et al. 2016), such as the brown spruce longhorn beetle (*Tetropium fuscum*), present additional threats to tree growth and survival that could exacerbate impacts of SBW (MacDonnell et al. 2020). Fourth, post-SBW interventions such as salvage logging can alter ecosystem components such as dead wood availability, nutrient cycling, and succession (Lindenmayer et al. 2008). Collectively, such anthropogenic changes are likely to increase the uncertainty for both short-term and especially long-term resilience of forest ecosystems under SBW outbreaks. In addition, ecosystem integrity is viewed across long periods (a tree generation or longer), but outbreaks will temporarily (in the short term) affect ecosystem characteristics and services, especially at the local scale. Relative to the lifespans of people, some of these impacts can be significant, even if they fall within the range of natural variability, and should be factored into any risk assessment.

Ecosystem properties, functions, and services under SBW outbreaks

A. Stand dynamics

As a major forest disturbance, SBW outbreaks drive stand successional trajectories through impacts on tree growth and mortality (Blais 1983; Bouchard et al. 2006; Martin et al. 2019), with cascading effects on many aspects of biodiversity and ecosystem functioning supported by these stands. Conversely, the severity of defoliation and degree of tree mortality are influenced by stand composition and structure (MacLean 1980; Berguet et al. 2021), resulting in a high degree of spatial heterogeneity, particularly in mixedwood forests. Region-wide outbreak suppression is expected to remove or reduce the natural, periodic imprint of SBW outbreaks not only on the age structure, composition and regeneration of individual stands, but also on forest heterogeneity on the landscape, with poorly understood consequences on forest ecosystems. Although salvage logging aims to mitigate the impacts on timber supply, it has also been shown to elevate SBW defoliation on spruce regeneration (Cotton-Gagnon et al. 2018) (see also section on Management – Pre-emptive harvest and salvage logging, AS 14).

SBW outbreaks have long been an intrinsic component of forest dynamics; stands are generally resilient across longer timescales, although the immediate successional and ecological effects may be pronounced and vary with pre-outbreak stand characteristics (Sanchez-Pinillos et al. 2019). Widespread mortality of balsam fir and, to a lesser extent, of spruce may initially convert softwood stands into mixedwood (Virgin and MacLean 2017). These successional trajectories, however, can be further shaped by stand management or, in some regions, interact with severe moose browsing (Leroux et al. 2021). Parts of the boreal forest, which have recently begun to experience novel outbreaks, may shift from black spruce-dominated forests to more open habitats (Pureswaran et al. 2015), with significant effects on biodiversity, nutrient cycling, and carbon sequestration. In contrast, boreal spruce forests with a history of SBW outbreaks show a high degree of resilience (Sanchez-Pinillos et al. 2019), although climate change introduces a significant degree of uncertainty (see section on Consequences of climate change, AS 4). Across much of the managed spruce-fir forests, the natural role of SBW in stand dynamics, thinning, and regeneration has diminished with more intensive forest management and salvage logging, and these factors need to be considered jointly in assessing the impacts of outbreaks and their management on ecological communities and ecosystem processes.

B. Biodiversity

Although biodiversity is not an ecosystem service in and of itself, it underpins and can be used as an indicator of the structure, functioning, resilience and ultimately integrity of ecosystem services provided by forests (Brockerhoff et al. 2017). The cumulative biodiversity of a forest represents the sum of supporting and regulating services such as natural pest control, nutrient cycling, carbon sequestration, water purification, and pollination. There are limited data available to examine biodiversity responses to a budworm outbreak. Here we discuss the more general biodiversity responses for taxonomic groups where data are available.

One of the best-known biodiversity responses to budworm are increases in abundance in the forest bird community. A short-term (1–5 years) increase in numbers of forest birds begins prior to obvious defoliation and lasts several years, presumably associated with a pulse in food availability in the form of SBW in the fifth and sixth instars and pupae (Mook 1963; Venier et al. 2009; Venier and Holmes 2010; Germain et al. 2021; Moisan Perrier et al. 2021). There is evidence of both a community-wide numerical response as well as shifts in foraging behaviour to take advantage of budworm abundance (Venier and Holmes 2010). Additionally, there are very large short-term increases in a select few budworm-linked species (bay-breasted warbler, *Setophaga castanea*; Tennessee warbler, *Oreothlypis peregrine;* and Cape May warbler, *Setophaga tigrina*) (Venier et al. 2009, 2011; Venier and Holmes 2010; Moisan Perrier et al. 2021; Germain et al. 2021), at both local and regional scales (Drever et al. 2018). It is unclear how important these bird population pulses are to the long-term dynamics and viability of populations, but a steady decline in populations of all three budworm-linked species during the inter-outbreak period from 1990 to 2010 (USGS https://www.mbr-pwrc.usgs.gov/) suggests that outbreaks may support long-term viability of these populations. Spruce budworm outbreaks also lead to significant bird habitat alteration (Holmes et al. 2009; Venier and Holmes 2010). Although defoliation and tree mortality reduce foliage for foraging as well as nesting sites for canopy nesters, and cover from predation, the associated pulse of deadwood can provide foraging and nesting opportunities for bark gleaners and cavity nesters, such as woodpeckers (Vaillancourt et al. 2008; Venier and Holmes 2010; Craig et al. 2019).

Through their impacts on canopy and tree mortality, SBW outbreaks trigger changes in the understory vegetation and associated biodiversity. In a comparison of understory vegetation recovery after logging, fire, and budworm outbreak, logging promoted the rapid expansion of tall shrubs, whereas SBW led to lower shrub cover through slower opening of the canopy coupled with retention of non-host trees (Kemball et al. 2005) relative to SBW. The high shrub coverage following logging resulted in more homogeneous stands with lower diversity compared with natural disturbances, with negative consequences for conifer recruitment and future stand composition and productivity.

The pulse of deadwood also supports the naturally occurring guild of saproxylic beetles that contribute to decomposition of deadwood and to nutrient cycling and act as a key food source for birds and mammals (Ulyshen 2016). Salvage logging following stand mortality due to SBW greatly reduces the quantity of fresh deadwood, which in turn alters the saproxylic insect community (Norvez et al. 2013). In Scandinavia, where the boreal forest has been managed intensively (Niemelä 1999; Stenbacka et al. 2010), the amount of dead wood has dramatically decreased, resulting in significant impacts on many saproxylic organisms (Kaila et al. 1997; Grove 2002; Stenbacka et al. 2010). Salvage logging alters stand structural complexity and ecosystem functioning as well as plant and animal community structure (Belle-Isle and Kneeshaw 2007; Lindenmayer et al. 2008; Cobb et al. 2010; Leverkus et al. 2021; McNie et al. 2023). In a meta-analysis of the effects of salvage logging on biodiversity across 24 species groups, saproxylic species were especially negatively affected (Thorn et al. 2018). Forest beetle diversity at larger scales may be enhanced through increased host tree heterogeneity when treated with biological insecticides like *Btk* (*Bacillus thuringiensis* var. *kurstaki*) (Wayland et al. 2015).

Some species, including species at risk as well as economically, culturally, or environmentally important species that are associated with forests prone to SBW outbreaks, are of particular concern. Although adapted to these disturbances, many of these species already face a significant number of primarily anthropogenic threats that, individually and cumulatively, have brought their populations to the limits of, or below, self-sustaining levels. The additional impact of habitat change from a widespread SBW outbreak may reduce local or regional populations and impair critical habitat connectivity. Targeted intervention to prevent the SBW-caused habitat decline may be required to preserve populations of species associated with mature spruce and fir forests. For example, economically and ecologically important cold-water fish species (e.g., Atlantic salmon, *Salmo salar*, and brook trout, *Salvelinus fontinalis*) are particularly sensitive to changes in water quality and temperature. Stand mortality caused by a SBW outbreak, or large-scale salvage logging in response, may elevate water temperatures and sedimentation, affecting their habitat. Another example is the Bicknell's thrush (*Catharus bicknelli*), a threatened species in Canada that breeds in high-elevation, young, dense, coniferous forests regenerating from disturbance (Sprugel 1976; Nixon et al. 2001; Townsend et al. 2020b). The species has a limited range and is threatened by activities that reduce stem density (e.g., pre-commercial thinning), and by an increase in hardwoods predicted by climate change (Environment and Climate Change Canada 2020b). A SBW outbreak resulting in the simultaneous loss of young, dense coniferous forests across the species limited range could jeopardize all its breeding habitat. Conversely, suppression of this natural, stand-replacing SBW disturbance could reduce the availability of dense, regenerating forests in the future.

C. Nutrient cycling

Through a suite of changes associated with SBW defoliation—litterfall, canopy openness, tree growth and mortality, coarse woody debris, and stand dynamics—outbreaks have significant consequences for biogeochemical processes in mixedwood and boreal forests on both shorter- and longer-term timescales. Yet, these effects have received surprisingly little attention until recently and represent a major gap in our understanding of how this iconic defoliator affects nutrient fluxes, soil carbon, microbial function in the rhizosphere, and the hydrological cycle. Much of our current knowledge of biogeochemical consequences of insect outbreaks come from other defoliators (e.g., Lovett et al. 2002; Grüning et al. 2017), and from bark beetles (especially mountain pine beetle, e.g., Edburg et al. 2012), the latter of which differ from SBW in several key aspects, including the feasibility of preventing an outbreak with EIS. It is known, however, that SBW outbreaks lead to increased nutrient leaching (Houle et al. 2009) and, at least in the case of the western spruce budworm, altered throughfall chemistry and nitrogen and phosphorus uptake (Arango et al. 2019). Most recently, a study from boreal forests in Quebec (De Grandpré et al. 2022) found that as both nutrient fluxes and uptake in outbreak-affected stands increased, so did the foliage nutritional quality of the surviving host trees—with implications for bottom-up ecosystem feedbacks driving SBW population dynamics.

Relatively little is known about microbial and mycorrhizal responses to SBW defoliation, although these are likely to mediate the most immediate shifts in decomposition and nutrient mineralization, with cascading effects on other ecosystem processes, including carbon sequestration (Lovett et al. 2002). Shifts in the soil abiotic environment and stand composition due to tree mortality, especially towards increased hardwood content, should produce major changes in nutrient dynamics; however, these effects may be dramatically modified by biomass removal, including salvage logging (Thiffault et al. 2011, but see Martineau et al. 2020). We also know little about how SBW impacts on biogeochemical processes in forest soils cascade down to changes in water chemistry and the food webs of forest streams and lakes, although research from other outbreaking insect species suggest that forest defoliators can be major drivers of nutrient cycling in aquatic ecosystems (Woodman et al. 2021). On longer timescales, climate change is also predicted to increase the uncertainty in many biogeochemical processes, particularly for boreal forests.

D. Carbon sequestration

Forests play a vital role in the global carbon cycle, storing and releasing it in dynamic equilibrium. SBW outbreaks can have significant effects on forest carbon (C) budgets through damage to photosynthetic tissues (i.e., needles), reduction in tree growth and increased tree mortality (Kurz et al. 2008; Dymond et al. 2010; Liu et al. 2018). Research suggests that there is a lack of accurate methods for C budget quantification with respect to insect disturbances (Liu et al. 2018). The Carbon Budget Model of the Canadian Forest Sector (CBM–CFS3) has been used to simulate and forecast ecosystem C dynamics affected by SBW disturbances (Kurz et al. 2008; Hennigar and MacLean 2010). Such studies found that a decrease in NEP may convert boreal forests from C sinks into C sources after SBW outbreaks. The CBM–CFS3, however, is an empirically driven model rather than a process-based model, which limits its ability to forecast future outcomes under novel conditions. There is ongoing work on a hybrid model (TRIPLEX-Insect) that uses both empirical and mechanistic approaches to quantify forest mortality and C sequestration in response to SBW outbreaks at the stand and regional scales (Liu et al. 2018, 2019b, 2020). TRIPLEX-Insect predicts a potential conversion of forests from C sinks to sources under SBW disturbance; however, a rapid recovery trend in annual NEP was found if defoliation was reduced, suggesting that affected forests may return to acting as a C sink faster if defoliation intensity is not severe and long-lasting. TRIPLEX-Insect was used in quantitative analysis of the effects of SBW outbreak and *Btk* treatments on the carbon dynamics in balsam fir, spruce, and mixed spruce-fir forests (Liu et al. 2019b, 2020) and showed that treatments can be effective in maintaining carbon in conifer forests during SBW disturbances, but that efficacy is dependent on host species and stage of outbreak. Most recently, TRIPLEX has been used to assess various scenarios of foliage protection and the EIS, for estimates of carbon dynamics (Liu et al. 2022). This will be an essential tool in examining the trade-offs of SBW management on ecosystem services.

Conifer and mixedwood forests of boreal and temperate regions represent a significant portion of global soil C capacity (Lal 2005), as the overstory only accounts for 15% of C storage in forests (Malhi et al. 1999). Because of their economic importance to the forest industry, reductions in wood volume and growth due to SBW defoliation are readily quantified (Dymond et al. 2010), and SBW impact mitigation through salvage logging has been shown to increase net C emissions for at least 10–20 years (Gunn et al. 2020). In contrast, the fate of forest C stock stored in the roots and soils is poorly understood in the context of SBW outbreaks. These C dynamics are mediated by soil biota, especially ectomycorrhizae that link C sequestration and nutrient cycling (Clemmensen et al. 2013). SBW defoliation is predicted to alter the complex interplay of fluxes, increase C inputs into soil via frass and litterfall, reduce photosynthate subsidies to ectomycorrhizae, and increase respiration from faster decomposition in warmer soils under the thinning canopy. The shifts in these fluxes, however, and the role of the soil biota mediating them are only now beginning to be examined. Modelling studies have shown that SBW outbreaks convert the affected forests from C sinks to sources (Dymond et al. 2010; Liu et al. 2019b), but the contribution of soil microbial responses to regional C sequestration remains overlooked. This gap may be critical for accurate forecasting of carbon budgets and mitigation under climate change, especially in boreal regions with large soil C pools previously unaffected by SBW outbreaks, and for linking carbon cycles to other ecosystem processes and forest management

Uncertainty

1. High uncertainty regarding how responses of forest ecosystems under a SBW outbreak interact with the cumulative effects of climate change, intensive forest management, and introduced species.
2. High uncertainty that changes in landscape heterogeneity and forest management have significantly increased SBW outbreak extent and severity and, therefore, decreased mature forest habitat provision.
3. High uncertainty in the resilience of species at risk associated with forests affected by SBW outbreaks or by salvage logging.
4. Moderate uncertainty in the contribution and responses of soil biota that mediate carbon sequestration and nutrient cycling to SBW, particularly in areas of novel outbreaks.
5. Moderate uncertainty that forests will be resilient to a SBW outbreak in the current range of SBW across large (regional) scales and long (across multiple tree generations) time frames.
6. Moderate uncertainty in how ecosystems will respond to salvage logging following a SBW outbreak.
7. Low uncertainty in short-term changes in ecosystem services but moderate uncertainty in the long-term changes and in areas of novel outbreaks.
8. Low uncertainty that biodiversity changes in response to SBW outbreak in ways that are generally considered to be natural across the majority of the range of spruce budworm.
9. Low uncertainty that SBW outbreaks will result in novel habitat conditions and disruption of ecosystem processes in areas north of current SBW outbreaks due to climate change.
10. Low uncertainty that a SBW outbreak will generate pulses of deadwood that support saproxylic communities and the biodiversity of some groups (e.g., bird communities).

Reducing uncertainty

Knowledge of past outbreaks and the resultant changes to ecosystem services (Mook 1963; Norvez et al. 2013) can provide a framework for formulating hypotheses about future effects and risks. Research on potential future impacts is needed using current information on forest composition in combination with succession predictions under climate change scenarios (Taylor et al. 2017) and should examine changes to the spatio-temporal aspects of both the disturbance and the recovery of the ecosystem. The following research objectives prioritize knowledge gaps to help improve our understanding of how forests and the ecosystem services they provide respond to SBW outbreaks:

1. Quantify the long-term impacts of SBW outbreak suppression on forest ecosystems that have evolved under a SBW disturbance regime in the context of climate change.
2. Assess the change in severity and spatial extent of SBW outbreaks due to anthropogenic disturbances including climate change, forest management, and introduced species.
3. Quantify the compound impacts of non-native species and SBW on tree mortality/susceptibility.
4. Quantify the effects of SBW outbreaks on a broad suite of biodiversity across the spatial extent and temporal cycle of an outbreak with and without salvage logging.
5. Assess the impact on ecosystem of responses to SBW under recent shifts in the distribution and dynamics of outbreaks under climate change.
6. Characterize the soil biota, carbon and nutrient fluxes, and stream responses to defoliation and tree mortality.
7. Determine the interaction of introduced species with SBW to shape post-outbreak stand dynamics.

AS 8: SBW outbreaks will significantly affect cultural services of critical importance to rural and Indigenous communities in managed and protected areas (By J.J. Bowden, D. Churchill, C.C. Sponarski, M. Stastny)

Evidence

Key points:

- Forests provide a plethora of cultural ecosystem services that are often overlooked or inadequately captured in ecological and economic assessments of SBW impacts.
- These services are of immense importance to Indigenous and other rural and remote communities who live in or near areas that experience or have recently begun to experience SBW disturbance, as well as to other Canadians who visit these areas for recreation.
- Rural and remote communities represent nearly 20% of Canada's population and nearly 30% of Canada's GDP, largely supported by natural resources.
- Forest disturbances by insects, in some cases compounded by climate change, can alter forest ecosystems and the way people interact with rural landscapes.
- SBW outbreaks may shift in their intensity or geographic distribution due to anthropogenic drivers such as climate change, and protected areas that have not experienced this level of disturbance previously may also become affected. In these areas, the cascade of ecological effects triggered by novel outbreaks is more difficult to predict but is likely to lead to more severe or long-lasting consequences on cultural services.

Details:

Spruce budworm outbreaks are known to affect the provisioning services (e.g., timber; MacLean et al. 2002) as well as the regulating/supporting services (e.g., carbon sequestration and nutrient cycling; Dymond et al. 2010; Arango et al. 2019) provided by forests, and because such services are more readily quantifiable (Brockerhoff et al. 2017; Binder et al. 2017), they tend to be the focus of forest pest risk assessments. Forests, however, also provide a plethora of cultural ecosystem services that are often overlooked or inadequately captured in the ecological and economic assessments of SBW impacts. The distinction between what is traditionally known as non-consumptive non-timber forest products (Davidson-Hunt et al. 2001) and those services that involve hunting, wild foods) blurs the lines between provisioning and cultural values for rural and remote communities. This complexity highlights some of the challenges in properly accounting for the role cultural ecosystem services play in the socioeconomic value of forests (see section on Impacts of outbreaks, AS 6), and in capturing the diversity of stakeholders and Indigenous peoples and their values. For instance, cultural ecosystem services can also be viewed as "…the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences…" (Alcamo et al. 2003). Unfortunately, the intrinsic intangibility of some of these services, as compared to others, makes them challenging to quantify (Alcamo et al. 2003; Binder et al. 2017), but they are measurable (Hernández-Morcillo et al. 2013). Still, these services are of immense importance to Indigenous and other rural and remote communities (e.g., Burger et al. 2008; Duchesne and Wetzel 2002; Morzillo et al. 2015; Lyver et al. 2017) who live in or near areas that experience or have recently begun to experience SBW disturbance, as well as to other Canadians who visit these areas for recreation. Therefore, it is imperative to consider their valuation among different stakeholders and Indigenous peoples, and the risks and uncertainties surrounding their maintenance under SBW outbreaks. In this section, we highlight three distinct sources of cultural ecosystem services: rural and remote communities in general, Indigenous communities, and protected areas.

Rural and remote communities represent nearly 20% of Canada's population and nearly 30% of Canada's GDP (Infrastructure Canada 2019), largely supported by natural resources (Vodden and Cunsolo 2021). These natural resources drive rural and remote economies, but also form the basis of the cultural ecosystem services relied upon by these communities and attract visitors through tourism and outdoor recreation. Forested landscapes represent important cultural areas that enhance quality of life and provide individuals living in rural and remote communities a sense of place and connectedness to the land (Vodden and Cunsolo 2021); however, the distant, urban populations who may visit these areas for recreation also share some of these benefits and values. Activities such as hunting support a sizable economic sector of these regions (e.g., Condon and Adamowicz 1993), sustain local food security, support tourism, and contribute to the regional cultural identity (e.g., Department of Environment and Conservation 2015).

Forest disturbances by insects, in some cases compounded by climate change, can significantly alter forest ecosystems and the way that people interact with rural landscapes (e.g., McFarlane and Witson 2007; Jansson et al. 2015). These impacts may have significant consequences for some people; for example, tree mortality brought on by outbreaks of native defoliators such as SBW in Canada (Labadie et al. 2021) and geometrid moths in Norway (Jepsen et al. 2013) can promote good, early succession habitat for moose (Labadie et al. 2021), a prized species for hunting. These outbreaks, however, may also lead to ecological cascades that result in declines of important cultural species such as caribou (Jepsen et al. 2013; Serrouya et al. 2017; Labadie et al. 2021). Therefore, multiple stakeholders, Indigenous peoples, and socioeconomic values may need to be considered in anticipating how a SBW outbreak or, conversely, its mitigation may affect local or regional cultural services. These rural perspectives have implications for local economies and the provisioning of cultural and ecological goods and services (Jansson et al. 2015).

Indigenous Peoples are part of Canada's rural and remote communities, and, for millennia, forests have been important spiritual and recreational places (Lewis and Sheppard 2005; Marquina-Márquez et al. 2016). Within Indigenous communities and governments there is great local history and, in that history, great knowledge (Davidson-Hunt and O'Flaherty 2007; Asselin 2015). It is imperative to employ such place-based knowledge in understanding the interactions between people and resources, and ultimately in the development of successful nature-based solutions (including treatment programs for SBW). Although Indigenous peoples have deep traditional reliance upon the ecosystem services provided by forests (Duchesne and Wetzel 2002), they are also increasingly economic partners and associates in the forest sector through sustainable forest management (National Aboriginal Forestry Association 2021; Natural Resources Canada 2021). Forest management systems, however, are often centred on timber supply with an industry focus, which can create barriers to the proper incorporation and valuation of Indigenous forest values or ecosystem services (Bélisle and Asselin 2021; Caverley 2009).

Throughout history, SBW has acted as the dominant natural disturbance agent shaping the structure and function of spruce-fir forests in eastern Canada. Despite this, the lack of documentation reflecting an Indigenous perspective on this disturbance and its significance for cultural services is a major impediment to integrating naturalized knowledge into informed management of SBW outbreaks. Recently, this concern was highlighted and included as a key recommendation in a report of the Standing Committee on Natural Resources (2019) from the House of Commons. In comparison, the Indigenous Sámi people of northern Fennoscandia offer valuable accounts of herbivory of mountain birch forests by the native irruptive geometrid moths and by reindeer (e.g., Vuojala-Magga and Turunen 2015). This deep place-based knowledge dates back centuries of interactions that influence not only provision of services (e.g., timber), but also how far the Sámi had to move reindeer for grazing following severe defoliation in outbreak areas. These disruptions to livelihood were transient, lasting 4–10 years. In recent decades, however, the Sámi describe the growing incidence of outbreaks beyond their historical distribution as "biological death" and a "poison" to the land (Vuojala-Magga and Turunen 2015). This account highlights how climate change, traditional human practices and irruptive defoliators can interact to profoundly affect the cultural landscape in a region undergoing ecosystem change, but it also underscores the need for better information on Indigenous perspectives in Canada.

Protected areas in Canada intend to conserve unique ecosystems; as such, they are primarily located in rural and remote areas of the country (Environment and Climate Change Canada 2020a). In addition to their role in the conservation of Canada's natural heritage and its ecological integrity and habitats, protected areas foster regional tourism and provide important cultural services such as recreational assets (Scott and Lemieux 2007). These areas often have unique mandates with respect to allowing natural disturbances that seek to maintain ecosystem integrity, such as outbreaks of native defoliators (e.g., Parks Canada 2021). Protected areas may therefore be excluded from forest pest management programs, even if these decisions may have efficacy implications for regional pest management of SBW, especially under the EIS. During unmitigated outbreaks, widespread tree damage and mortality associated with SBW can be expected to negatively affect recreation in protected areas. Indeed, climate change can drive insect outbreaks that can reduce the use of recreational areas such as campgrounds, which in turn have economic consequences (Ayres and Lombardero 2000). During the short to medium term, the consequences on the provision of cultural services can range from direct (e.g., closure of trails and campgrounds due to falling tree hazard and liability concerns, and lower visitation due to a changed aesthetic, etc.) to indirect (e.g., reduced tourism in surrounding communities). Spruce budworm outbreaks may shift in their intensity or geographic distribution due to anthropogenic drivers such as climate change, and protected areas that have not experienced this level of disturbance previously may also become affected. In these areas, the cascade of ecological effects triggered by the novel outbreaks is more difficult to predict but is likely to lead to more severe or long-lasting consequences on cultural services. Finally, protected areas have a unique mandate and cultural value that may warrant management considerations, in comparison to the vast tracts of unmanaged boreal forest in eastern Canada where unmerchantable timber underlies the default strategy of "no intervention".

In the face of anthropogenic change, Indigenous, rural, and remote communities are important for understanding and setting sustainable forest management objectives, including those that explicitly address or prioritize cultural ecosystem services. Climate change is influencing, and will continue to influence, forested landscapes, including through the frequency, distribution, and severity of insect outbreaks (Price et al. 2013). Climate change-mediated insect disturbance is the most commonly reported disturbance pathway in forest ecosystems, a trend that is likely to continue (Seidl et al. 2017) and shift geographically. The recent geographic expansion of irruptive forest defoliators in boreal forests presents novel challenges to communities previously unaffected by such disturbances (Jepsen et al. 2013; Pureswaran et al. 2015). As the pressures on the integrity and maintenance of provisioning and regulating/supporting services in the face of SBW outbreaks increases under climate change, cultural services relied upon by rural and remote communities are also at risk. Indeed, SBW population growth rates are projected to increase dramatically at northern latitudes under future climate scenarios (see section on Consequences of climate change, AS 4), thus exposing new areas to unprecedented disturbances by this irruptive defoliator. As the intensity and distribution of SBW outbreaks shift under climate change, consequences for the forest ecosystems and their biodiversity can cascade down to cultural ecosystem services, such as culturally important species (e.g., caribou), potentially requiring targeted mitigation. Indigenous peoples are essential to the successful development of nature-based solutions (Townsend et al. 2020a) to forest disturbances under climate change. Finally, protected areas vulnerable to SBW are particularly susceptible to climate change impacts because they often include ecosystems that are at climatic extremes (e.g., high elevation/latitude) and restricted in space (Gonzalez et al. 2018). Moreover, ecosystems that are no longer in equilibrium with prevailing climatic conditions may then transition, through disturbance, to a new state that is entirely different from the previous one. As climate change continues to threaten the integrity of ecosystem services, it is critical to rural and remote communities, including Indigenous ones, that we understand the potential consequences and implications of large-scale forest disturbances for cultural values and provisions on the landscape.

Uncertainty

1. High uncertainty about quantifying the specific cultural services provided to rural and Indigenous communities.
2. High uncertainty about how climate change will interact with SBW outbreaks to affect the cultural services provided by forests.
3. High uncertainty in how Indigenous communities perceive irruptive forest defoliators.
4. Moderate uncertainty that unmitigated SBW outbreaks will negatively affect cultural services.
5. Moderate uncertainty in how Indigenous communities perceive treatment programs (i.e., aerial applications of insecticides).
6. Moderate uncertainty in the short-term integrity of cultural services provided by protected areas under SBW outbreaks.

Reducing uncertainty

1. Identify Indigenous perspectives/history in relation to irruptive forest insects, and their role in ecosystem services.
2. Exchange and co-create knowledge about Indigenous and remote community forest values affected by irruptive forest insects (and in light of climate change). Examine inherent differences among stakeholders and Indigenous peoples in the valuation and stewardship of ecological function and ecosystem services provided by forests.

3. Improve understanding of cultural services affected—or facilitated—by SBW outbreaks or their prevention, especially in rural and remote communities.
4. Quantify and determine the value of specific cultural ecosystem services provided to rural and remote communities.
5. Understand how climate change will interact with SBW outbreaks to mediate availability of cultural services.
6. Assess SBW impacts on protected areas and their provision of cultural services.

Management - Monitoring tools

AS 9: Current monitoring methods and intensities are adequate for effective management of SBW (By R.C. Johns, D. Carleton, J.P. Brandt)

Evidence

Key points:

- For the FPS, current monitoring techniques meet provincial SBW management needs and have been used for several decades by these provinces.
- For the EIS, intensive monitoring of L2 densities has permitted the effective detection and subsequent suppression of "hotspots" in Atlantic Canada.

Details:

The EIS and the FPS use the same proxies for assessing SBW population density, including overwintering second instar larvae (L2) density, defoliation, and pheromone trap captures (see Table 1 for a summary of SBW monitoring techniques applicable to both pest management and research). For the FPS, current monitoring techniques meet provincial SBW management needs and have been used for several decades by these provinces. Under the FPS, annual estimates of previous defoliation provide the basis for identifying stands vulnerable to tree mortality if protection is not deployed. Subsequent assessments of L2 density are then used to further pinpoint locations needing protection. The thresholds used for the FPS tend to vary somewhat among provinces but generally favour stands with multiple years of severe defoliation (~95%) and L2 densities likely to cause comparable damage the following year. Moreover, the FPS tends to focus on managing mature, high-value stands and plantations only that are soon to be harvested and assigns a lower priority to areas that are not commercially important. Finally, monitoring intensity is far less than that required for an EIS.

Table 1 – Part 1. Summary of monitoring techniques for various SBW life stages used to meet information needs for management and study (egg sampling, second instar larvae, large larvae). Modified and updated for both a Foliage Protection Strategy and the novel Early Intervention Strategy from information presented by Sanders (1980) and Dorais and Kettela (1982) and input from current and former provincial pest management staff (personal communication, Pierre Therrien, Dan Rowlinson, Dan Lavigne, Nelson Carter).

Table 1 - Part 1	Egg sampling	Second instar larvae	Large larvae
Key reference(s)	Morris (1954, 1955); Webb et al. (1956); Prebble (1975)	Miller and McDougall (1968); Miller et al. (1971); Miller and Kettela (1972); Hartling (1994, 2000); O'Shea (2004)	Atwood (1944); Morris (1955); Régnière & Sanders (1983); Régnière et al. (1989)
Purpose	Forecasting subsequent year populations and levels of expected defoliation. Track population trends numerically and spatially. Maps prepared to illustrate the extent and severity of infestation, and to identify areas in need of protection.	Forecasting subsequent year populations and levels of expected defoliation. Track population trends numerically and spatially. Maps prepared to illustrate the extent and severity of infestation, and to identify areas in need of protection.	Evaluate efficacy of insecticides, or develop life tables. Not used as a standard operational population survey method, but counts are included as part of pre- and post-spray population assessments to determine efficacy or larval mortality due to insecticide treatment (L6-pupal counts).

Table 1 - Part 1	Egg sampling	Second instar larvae	Large larvae
Sample unit and sampling intensity.	One whole branch from mid-crown of a dominant or co-dominant host tree. Data expressed as numbers of egg masses per 10 m^2 of foliage (originally per 100 ft^2). Different methods of determining area of foliage have evolved in different parts of Canada (see Sanders 1980) but generally it is the length of branch from tip to last living branch nearest tree bole (= L) x width (= W) at 1/2 L or the maximum width of the branch. Number of plots in given area varies depending on purpose of survey and jurisdiction: 1 plot/50 000 ha for extensive surveys in Ontario; or from a low of 1 plot/ 10 000 ha in Nova Scotia to a high of 1 plot/1 000 ha in Ontario for potential treatment areas. Sequential sampling dictates sampling intensity within plot. In practice, actual sequential sampling (i.e., re-visiting plots to sample more trees) was not rigorously followed due to pragmatic considerations and cost. Using sequential table, tallying egg masses could be stopped when number of egg masses fell into either "Low" or "High" categories (and tally was recorded); where tally did not fall in either of latter categories and more sampling was indicated, they were classified as "Medium".	One whole branch taken from the mid-crown of a dominant or co-dominant host tree is used everywhere except in Quebec, where a 45 cm branch tip is the sample unit. A total of 3 branches per tree species per plot are collected in Quebec, the Maritimes and Newfoundland, and a minimum of 10 in Ontario. Data are expressed as numbers of second instar larvae per unit of branch area (length × width) of 10 m^2. Variance-mean relationships are provided by the references to determine the number of samples required for a prescribed level of accuracy. Depending on the jurisdiction, data are also expressed as the average number of larvae/branch.	45 cm branch tip from mid-crown of the host tree is used currently. Initially the 45 cm branch tip was used for extensive surveys and the whole branch was used for intensive surveys. Variance-mean relationships are provided by the references to determine the number of samples required for a prescribed level of accuracy, or sequential sampling can also be used. Pre- and post-spray population estimates were expressed as the number of larvae/branch or number of larvae/shoot. The latter was used when defoliation estimates were made using the Fettes method.
Summary	This technique was in widespread use during the 1950s outbreak up until it was replaced with the L2 survey in the 1980s (c. 1985). Infestations in balsam fir are classified on basis of sequential egg sampling carried out on a single mid-crown branch, as either light, moderate, or severe. Same classification is used for white spruce, on assumption that statistics derived for balsam fir can be applied to white spruce (this assumption is untested). For extensive egg surveys, maximum number of branches sampled at each sample plot is five in Quebec, three in New Brunswick, and six in Ontario. Foliage on sampled branches is examined visually for all egg masses and counted. When data are used to express population density, averages are determined by summating egg counts per sample plot, then dividing the total branch surface examined.	During the 1970-1980s outbreak, this technique eventually replaced the egg mass survey. Overwintering SBW second instar larvae are liberated from their hibernaculae (as the latter is dissolved) and the larvae washed from the foliage of branch samples with a hot sodium hydroxide solution. Second instar larvae are then counted and expressed as larvae per 10 m2 of foliage. After soaking and agitation in the solution for a period of time, the rinseate is put through a filtration and extraction process and the larvae are finally saved on gridded filter paper to be counted under a microscope. Figure 3 in Miller et al. (1971) or Figure 1 (includes data for Ontario) in Dorais and Kettela (1982) define the relationships between expected defoliation of balsam fir branches and the density of eggs masses and second instar larvae.	Foliage is examined visually for presence of larvae, or else larvae are extracted by mechanical aids (i.e., drum technique) and then counted. On freshly gathered branches, third and fourth instar larvae are found only in buds or staminate flowers. A simple two-way classification into low or high density populations on balsam fir and red spruce by sequential sampling is given by Prebble (1975).
Thresholds	Varies somewhat by jurisdiction. See sequential sampling tables on pages 5-6 of Dorais and Kettala (1982).	See Figure 1 in Dorais and Kettela (1982) for relationships in NB and in ON.	See Table 7 of Sanders 1980.
Recent jurisdictional modifications (& rationale), effectiveness, uncertainties, gaps:			
New Brunswick (EIS)	Not used under the EIS.	Under EIS, both extensive (operational) and intensive (supplemental) L2 surveys are used to forecast subsequent year populations and levels of expected defoliation in order to identify "hotspot" areas for treatment. Also used for assessing efficacy of EIS treatment by comparing year-to-year changes in population density in treated and untreated plots. Sample unit is one 75-cm branch tip taken from mid-crown of dominant or co-dominant host tree. Total of three trees with one branch per tree are sampled. Hotspots are treated when average L2/branch is greater than six L2/branch. Weighted L2 values when combined with weighted values for susceptible hosts are used to calculate protection priority values. Depending on the protection value used and spray optimization conducted, areas with lower L2 values less than six L2/branch may be included.	NA
Newfoundland & Labrador (EIS)	Not used under the EIS.	Province-wide extensive & intensive L2 surveys. Monitoring for areas meeting the EIS threshold of greater than six L2/branch.	NA

Table 1 - Part 1	Egg sampling	Second instar larvae	Large larvae
Nova Scotia (EIS)	Not used under the EIS.	Province-wide extensive & intensive L2 surveys. Monitoring for areas meeting the EIS threshold of greater than six L2/branch.	NA
Quebec (FPS)	Not used any longer.	Branches taken from middle of top third of crown.	Branches taken from middle of top third of crown.
Ontario (FPS)	No formal egg mass sampling is conducted for operations; only used for research studies.	One upper mid-crown branch, 45 cm long, taken from each of 10 representative balsam fir trees. L2 larvae are counted, and these counts are used to provide a defoliation forecast of nil, light, moderate, or severe, for next growing season. The data are used to guide operational treatment programs where defoliation is forecasted to be moderate-to-severe.	Only conducted during operational foliage treatment programs to determine spray efficacy.

Table 1 – Part 2. Summary of monitoring techniques for various SBW life stages used to meet information needs for management and study (pupae, adults, defoliation, tree mortality, top-kill or breakage). Modified and updated for both a Foliage Protection Strategy and the novel Early Intervention Strategy from information presented by Sanders (1980) and Dorais and Kettela (1982) and input from current and former provincial pest management staff (personal communication, Pierre Therrien, Dan Rowlinson, Dan Lavigne, Nelson Carter).

Table 1 - Part 2	Pupae	Adults	Defoliation (aerial and ground assessments)	Tree mortality, top-kill, or breakage
Key reference(s)	Morris (1955); Prebble (1975)	Sanders (1986, 1988, 1996); Lyons et al. (2002); Hartling (2000)	Waters et al. (1958); Webb et al. (1956), Fettes (1950); Hartling (2000)	Dorais and Kettela (1982), Hartling (2000)
Purpose	Classifying population levels. Not used as a standard operational population survey method, but counts are included as part of post-spray efficacy assessment to estimate larval mortality due to insecticide treatment (L6-pupal counts).	Monitoring changes in low density populations, and trends in population density. Aids in identifying areas to conduct more intensive and costly L2 surveys, especially in conjunction with the EIS effort.	Determining extent and severity of current defoliation through aerial surveys. Useful for identifying areas to consider for insecticide treatment. Data used as input to the SBWDSS. Branch and tree-level defoliation estimates used for measuring success of treatment. Defoliation also determined from ground during pre- and post-treatment assessments. Various methods used - for example, ocular method used to rate current and previous defoliation. Fettes method used to rate current defoliation.	Estimating level of tree mortality during or soon after an SBW outbreak to determine need for salvage harvesting. Useful for calculating annual allowable cut scenarios.
Sample unit and sampling intensity.	45 cm branch tip is used for extensive surveys and the whole branch is used for intensive surveys. Sequential sampling dictates sampling intensity.	Adult male moths captured with pheromone-baited traps. Trap counts are regarded as a relative index of population density and have the most value when the same plots are used each year and the sampling distribution is not too sparse. Concerns have been expressed about year-to-year consistency of the lure, type of trap to use, and number of traps per plot.	Parallel flight lines at spacing of 2-13 km apart depending on jurisdiction and data requirements for survey. For ground assessments, ocular or Fettes method used to assess current defoliation on branch. Ocular method used to rate current and previous defoliation for tree.	Parallel flight lines at spacing of 5-10 km apart depending on jurisdiction and data requirements for the survey. Often completed simultaneously with defoliation surveys.
Summary	Technique similar to that for large larvae. Foliage is examined visually for presence of pupae, or else larvae are extracted by mechanical aids (i.e., drum technique) and then counted. A simple two-way classification into low or high density populations by sequential sampling is given by Prebble (1975).	This technique, developed in the 1980s and early 1990s, uses pheromone-baited traps to monitor changes in low populations. It is relatively efficient, effective, and easy to use as an early warning of populations changes or trends. There are some jurisdictional differences in type of trap used, trap placement, and number of traps per plot.	Aerial defoliation surveys are conducted using either fixed-wing or rotary-wing aircraft depending on the jurisdiction, the terrain, and purpose of the survey. Boundaries of visible defoliation are mapped onto conveniently scaled maps, again depending on purpose of survey. More recently, defoliation data are captured digitally on computer tablets using digital forest inventory maps. Overall purpose of aerial defoliation surveys are to map extent and severity of SBW defoliation (and tree mortality when it occurs).	Surveys are conducted using either fixed-wing or rotary-wing aircraft depending on the jurisdiction and the terrain. Areas of mortality are mapped onto conveniently scaled maps, which vary with jurisdiction. More recently, extent and intensity of mortality is captured digitally on computer tablets using digital forest inventory maps.

Table 1 - Part 2	Pupae	Adults	Defoliation (aerial and ground assessments)	Tree mortality, top-kill, or breakage
Thresholds	See Table 9 of Sanders 1980.	A catch of 100 moths corresponds to a density of 25 second instar larvae per 10 m^2 of branch surface area. Thus, this threshold is indicative of the need for more intensive larval sampling as appropriate. Because much of Sanders' work was done in boreal mixedwood forests of Ontario, there have been concerns about the applicability of this threshold in the Maritime Provinces where the fir content is higher. Thus, higher trap catches would likely be required than that of Ontario to reflect similar L2 branch densities.	In all regions of eastern Canada except Ontario, current defoliation is mapped and classified into three categories: light (up to 30% defoliation), moderate (31 to 70% defoliation), and severe (more than 70% defoliation). In Ontario, only the area of moderate to severe defoliation is mapped and shown as one category.	NA

Recent jurisdictional modifications (& rationale), effectiveness, uncertainties, gaps:

New Brunswick (EIS)	NA	Province-wide network of 100 semi-permanent sample plots (plots are moved if stands are harvested) where three pheromone traps are placed ~50 m apart within a plot, annually. Captured moths are counted and these data are used to monitor changes in low density populations, or trends in population density. Trap catch density and spatial distribution aid in identifying areas for general L2 sampling and/or for comparison to the EIS L2 threshold. The provincial program is augmented by about 50 plots monitored by forest industry.	The province has recently adopted a change-detection approach that relies exclusively on satellite imagery.	NA
Newfoundland & Labrador (EIS)	NA	Province-wide annual pheromone trapping survey to aid in selecting areas for L2 sampling related to EIS threshold. Also, subset of traps used for daily monitoring of trap catches to aid in identifying potential moth immigration events. The provincial program is augmented by the Budworm Tracker community science program, which adds 280 pheromone trapping locations.	Province-wide or directed annual defoliation survey.	NA
Nova Scotia (EIS)	NA	Similar to New Brunswick except that the provincial program is augmented by additional 180 pheromone trapping locations as part of the Budworm Tracker community science program.	Province-wide or directed annual defoliation survey.	NA
Quebec (FPS)	Not used operationally.	Used only for endemic populations.	Also, the province is using remote sensing for northern areas and eastern areas not currently sprayed.	No aerial survey. Use of remote sensing with ground truthing.
Ontario (FPS)	Not used operationally.	Pheromone baited traps monitor SBW population fluctuations using a pre-existing network of 80-100 spruce-fir monitoring plots a various locations, with 3 pheromone traps per plot.	Aerial surveys are flown mid-summer after peak defoliation, when red needles damaged by budworm larvae and frass collected in webbing are most visible. Flight lines are chosen based on where defoliation is known or expected to occur, as well as visibility and experience of staff. Polygons outlining areas of defoliaton are recorded using a digital mapping tablet with PC Mapper software. Use of satellite imagery to supplement data collection is also being investigated. Ground verification of defoliation severity and causal agent (i.e., SBW) is done post flight. Data are cleaned and made available through the Land Information Ontario (LIO) portal.	Annual tree mortality is recorded during the aerial defoliation surveys and reported as a separate forest disturbance event.

NOTE: Most methods and sampling regimes for SBW were largely developed when populations were at outbreak levels for research purposes. These methods were often put into practice operationally in a "modified" manner because of pragmatic and economic constraints.

The EIS, in contrast, requires methods for detecting locations with low but rising SBW populations (i.e., hotspots) and identifying priority areas for SBW control (MacLean et al. 2019; Johns et al. 2019). The basic protocol for defining treatment priority areas in the EIS involves: (1) detecting hotspots; (2) assessing forest susceptibility; and (3) selecting and assigning control tactics. The EIS detects and monitors hotspots through annual sampling of L2 density. These data provide the first consideration in prioritizing areas for treatment. L2 monitoring is useful for several reasons. First, L2 density acts as a fair proxy of population density relative to the irruption threshold where populations seemingly shift into their outbreak phase (Régnière et al. 2019a). Second, L2 monitoring occurs in the fall and winter, which at least in New Brunswick provides sufficient time for planning budgetary, treatment, and communication activities for the upcoming season. Finally, L2 monitoring has been the convention for monitoring annual SBW densities for the FPS since the early-mid 1980s (see Table 1). As such, the techniques and infrastructure for collecting and extracting larvae from branches are well established and relatively low cost (see references in Table 1).

Monitoring via L2 densities is much more intensive in the EIS than in the FPS, as the former requires a much higher spatial resolution to ensure that potential hotspots are not missed (MacLean et al. 2019; Johns et al. 2019). Although L2 monitoring as the primary tool for identifying treatment priority areas is probably not replaceable, secondary proxies of population density could be used to increase monitoring efficiency. These proxies could include pheromone trapping of male moths (Sanders 1988), defoliation surveys (MacLean and MacKinnon 1996), or insectivorous bird surveys (Germain et al. 2021; Moisan Perrier et al. 2021). Additional insights could be gained from modelling SBW flight behaviour, which could help identify locations where immigrant moths could potentially lay eggs (see section on Population dynamics and outbreaks, AS 3). Instances of abnormally high SBW activity may trigger a second round of L2 sampling in the affected area, which could further help to define treatment area boundaries and increase monitoring efficiency. All areas with known SBW L2 densities above the hotspot threshold (i.e., seven second instar larvae per branch) are set to high priority to ensure their treatment with insecticides. L2 monitoring efficiency in New Brunswick has been further enhanced through insights provided by one of several approaches: defoliation assessments using remote sensing, roadside assessments, or aerial observations (Rahimzadeh-Bajgiran et al. 2018). In the future, efficiencies could be further enhanced through integration with modelling tools to predict moth dispersal patterns via weather forecasting (Régnière et al. 2019b; Garcia et al. 2022) or radar (Boulanger et al. 2017).

Notably, although the FPS has been used historically for more than 60 years to manage SBW throughout eastern Canada, all our understanding of monitoring under the EIS to date is based on the current application of this strategy in New Brunswick. Whether the same monitoring methods, action thresholds, and monitoring intensity will remain consistent across other jurisdictions looking to use the EIS remains an open question and one that requires further investigation. In New Brunswick, Nova Scotia, and parts of Quebec (i.e., Lower St. Lawrence), current monitoring intensity may be adequate for the EIS. In New Brunswick, current monitoring intensity (~2,000 plots across a ~5 million ha area) appears to be adequate. It remains uncertain, however, whether New Brunswick could reduce plots and still achieve the same success. If remote sensing of defoliation can give adequate and reasonably precise measures to detect hotspots in unexpected places, this could be used to enhance monitoring and potentially extend the use of EIS in larger jurisdictions where intensive L2 monitoring is neither logistically nor economically possible. One issue with this approach is it will only capture sites that already have active populations but not those that have moved in at the end of that season. In this sense, using only remote sensing estimates of defoliation without information from L2 assessments could result in constantly missing the leading edge in expansion of hotspots.

Uncertainty

1. High uncertainty around the linkage between remote sensing estimates of defoliation and L2 density in the subsequent year.
2. High uncertainty about SBW moth flight biology for behavioural modelling of likely immigrant destinations and oviposition (e.g., how far and fast moths fly, take-off biology, etc.).
3. High uncertainty whether SBW flight modelling will have sufficient precision to project potential locations where immigrant moths could potentially lay eggs. (i.e., to guide subsequent L2 monitoring).
4. Moderate uncertainty about the necessity of using L2 monitoring as the primary basis for designating EIS treatment areas (versus other potential proxies from moth flight modelling and remote sensing).

5. Moderate uncertainty about the likelihood of consistency of the current EIS "hotspot" threshold across different jurisdictions.
6. Moderate uncertainty about the frequency and reach of mass moth dispersal events and the extent to which they contribute to outbreak spread at local and regional scales.
7. Low uncertainty about the efficacy of current EIS monitoring methods and survey intensity in New Brunswick.
8. Low uncertainty about the adequacy of monitoring intensity for the FPS in provinces not using the EIS (i.e., Ontario and Quebec).
9. Low uncertainty about the inadequacy of forest inventories in some jurisdictions for use in conjunction with L2 densities for determining treatment priority areas.

Reducing uncertainty

1. Assess and quantify optimal L2 monitoring intensity for different jurisdictions.
2. Determine, through both biological studies and modelling, the frequency and extent of mass moth dispersal events and how they influence outbreak spread and dynamics at local and regional scales.
3. Assess how moth flight biology is influenced by meteorological conditions at local and regional scales.
4. Improve forest inventory data in some jurisdictions to better represent the composition of forest stands.
5. Enhance understanding of what types or combinations of surveillance methods might be used to monitor remote locations in the context of both EIS and FPS (i.e., areas with no road access).

Management - Early Intervention Strategy (EIS)

AS 10: There are essential elements and principles of the EIS that must be included for a successful approach to managing SBW (By R.C. Johns, J.J. Bowden, E.R.D. Moise)

Evidence

Key points:

- The EIS is essentially a proactive, large-scale approach intended to prevent outbreak rise and spread by detecting hotspots and identifying priority areas for SBW treatment with insecticides.
- Although monitoring approaches are similar, sampling intensity and action thresholds used to define treatment areas in the EIS differ substantially from the FPS.
- The basic protocol for defining treatment priority areas in the EIS involves: (1) detecting hotspots early; (2) assessing forest susceptibility; and (3) selecting and assigning control tactics.
- Efficacy under the EIS entails using tactics that can "add" SBW mortality through insecticide application to that occurring naturally while having a limited impact on natural mortality agents.
- Other important elements of the EIS include cost-benefit analyses, and communications and outreach to ensure public support for the program.

Details:

The key elements of a successful EIS (conceptual framework, Fig. 5) have been articulated by Johns et al. (2019) and to some extent validated by MacLean et al. (2019). This framework was based, in part, on decades of theoretical and field-tested population management "best practices" used for vertebrate and invertebrate pests (Johns et al. 2019). This EIS framework provides the basis for guiding effective management, including methods to evaluate EIS efficacy, determining under what conditions it might work best, and identifying knowledge and technical gaps for future research.

A conceptual framework for the EIS (see Fig. 5) illustrates the relationship between its different components. The expectation of "contagious" outbreak dynamics provides the core ecological justification for the EIS. In turn, the aims of the EIS dictate monitoring and treatment prioritization protocols, population control practices and tactics, and the criteria used in cost-benefit analyses. These particular components are highly dependent upon one another, in that challenges or innovations in one component will likely influence the efficacy or feasibility of the others. Proactive communications and outreach are essential for disseminating information and garnering social licence to allow all other aspects of the strategy to operate (Johns et al. 2019).

Figure 5. A conceptual framework for the EIS illustrating the relationships between its different components.

1. <u>"Contagious" outbreak dynamics.</u> The EIS is based on an interpretation of SBW population dynamics asserting that outbreaks are triggered by external factors that drive populations to rise above natural constraints (i.e., irruptive threshold). As populations rise, moths immigrate and seed hotspots and thus drive the propagation of the outbreak. The EIS is essentially a proactive, large-scale approach to preventing outbreak rise and spread through finding and suppressing rising populations (i.e., hotspots) before they can establish and propagate further spread (see also section on Population dynamics and outbreaks, AS 1–3).

2. Hotspot detection and monitoring. The EIS requires efficient methods for detecting hotspots and identifying priority areas for SBW control. Many of the same proxies for population density used in the FPS are also used in the EIS (see Table 1, AS 9). The sampling intensity and action thresholds used to define treatment areas in the EIS, however, differ substantially. The basic protocol for defining treatment priority areas in the EIS involves: (1) detecting hotspots early; (2) assessing forest susceptibility; and (3) selecting and assigning control tactics (see also section on Monitoring tools, AS 9).
3. Population control. The EIS does not aim to eradicate SBW from the forest, nor is population control achieved simply by inflicting maximal mortality in low-density but rising populations. Populations respond in a variety of ways following insecticide treatments, and knowing how and why they respond in certain ways is the key to effective population management. In the ideal scenario, the EIS is able to impose sufficient additive mortality to reduce populations below the irruption threshold, below which natural mortality agents may keep populations low. This strategy contrasts with the FPS, which aims only to minimize herbivory and keep host trees alive by keeping populations low during the growing season with no expected impact on SBW populations in the subsequent year. Efficacy in the context of the EIS entails using tactics that can "add" mortality to that occurring naturally while having a limited impact on natural mortality agents. Treatments must also be applied across relatively large areas (i.e., area-wide management) such that immigrant moths from uncontrolled regions or jurisdictions cannot replenish treated populations (see also section on Population dynamics and outbreaks, AS 3).
4. Cost-benefit analyses. There are three main categories of cost-benefit analyses underlying the EIS. First, the economic feasibility of the EIS will ultimately depend on how management costs compare with potential losses associated with an uncontrolled SBW outbreak or with alternative strategies for management. Second, there are potential ecological costs (e.g., non-target impacts of insecticides; see section on Impacts of management on non-target species, AS 15) and benefits (e.g., preserving vulnerable habitat) that must be weighed. Third, there are important sociopolitical factors that need to be considered, including the ability of the EIS to treat on private and First Nations land, and government authority over Crown and protected lands (see also AS 5, 6, 7, 8, 15). These are likely to vary significantly across eastern Canada and would thus need to be evaluated in the context of each region or jurisdiction.
5. Communications and Outreach. Even if all aspects of the EIS work as intended and are highly cost effective, without public support the program could not be sustainable at levels needed for successful regional outbreak management. Controversy has long surrounded insecticide usage for SBW, and concerns linger to this day, despite the replacement of broad-spectrum chemical insecticides with more ecologically benign alternatives. A variety of audiences must be engaged and consulted before treatments can occur, including governmental decision makers, industry, environmental groups, Indigenous peoples, landowner organizations, provincial and federal parks, municipalities, provincial and local media, and residents. A core component of the EIS is a proactive (and bilingual) communications approach that improves knowledge about EIS science and management, SBW ecology, and the tactics used in the past and present. This proactive approach contrasts with the more reactive approach that has underpinned most past operations under the FPS. A few core principles guide the EIS communications approach. First, scientists and other experts are the communication ambassadors for EIS management and science. This approach gives audiences opportunities to engage with experts directly and reduces the likelihood of miscommunication around the underlying science or ongoing management efforts. Second, timely updates are communicated on all aspects of the EIS, including SBW population trends, the location and timing of treatments, and the ongoing progress in the management and scientific research. Finally, scientists and other experts address all public inquiries and concerns directly and openly, and, if possible, provide evidence using reference material from available scientific literature.

Uncertainty

1. Moderate uncertainty whether the core elements of the EIS can be applied effectively across all jurisdictions.
2. Low uncertainty that the core elements of the EIS reflect “best practices” and ecological principles underlying population management programs for myriad vertebrate and invertebrate pests.
3. Low uncertainty that the elements of EIS have been highly effective for controlling SBW outbreak rise and spread in New Brunswick to date.

Reducing uncertainty

1. Monitor and evaluate the long-term efficacy of the ongoing EIS in Atlantic Canada and Maine, USA.
2. Examine the application of EIS principles across a range of jurisdictions, locations, and local outbreak intensities.

AS 11: Sustained application of the EIS for the short term (~5 more years) will reduce the long-term impact of SBW in Atlantic Canada

(By J.J. Bowden, E.R.D. Moise, R.C. Johns)

Evidence

Key points:

- Currently, there is no evidence that the EIS applied in the short term will reduce long-term outbreak progression.
- There is compelling evidence that, in the short term, SBW populations are reduced by hotspot treatment, with SBW densities in northern New Brunswick consistently and substantially lower than in nearby Quebec (employing a different management strategy—the FPS).
- Long-term data will ultimately be required to determine the long-term impacts of the EIS in Atlantic Canada.
- Recent experience suggests that the EIS is an effective strategy for flattening the outbreak curve and preventing outbreak spread in Atlantic Canada with a much-reduced overall treatment area in New Brunswick in particular.

Details:

We currently have no evidence that the EIS applied in the short term will reduce long-term outbreak progression. We do, however, have compelling evidence published by MacLean et al. (2019) suggesting that the EIS is an effective approach for managing SBW in Atlantic Canada. After five years of EIS treatments to low but increasing populations, SBW densities in northern New Brunswick were substantially lower than in nearby Quebec, where a different management strategy (FPS) was used. Areas requiring treatment in New Brunswick increased from 2014 to 2018, but treated forests consistently showed reductions in population densities, and most required no treatment in the subsequent year. Several years of data since the MacLean et al. publication suggest that the EIS is an effective strategy for flattening the outbreak curve and preventing outbreak spread in Atlantic Canada.

Uncertainty

1. High uncertainty that sustaining the EIS for the short term will have benefits for the long term in Atlantic Canada (compared to alternative strategies).
2. High uncertainty around the direction, frequency, and intensity of mass dispersal events from outbreak areas and whether these will overwhelm EIS efforts in some or all adjacent jurisdictions (for more details see section Population dynamics and outbreaks, AS 3).
3. Moderate uncertainty that outbreak pressure in Atlantic Canada will subside when the Quebec outbreak collapses.
4. Moderate uncertainty whether changes in forest susceptibility since the end of the last outbreak will help suppress hotspot establishment and slow outbreak spread (e.g., eastern Nova Scotia forests are more susceptible, but forests in other parts of the province are more heterogeneous and less susceptible).

Reducing uncertainty

1. Monitor and evaluate the long-term efficacy of the EIS in Atlantic Canada.
2. Assess the extent to which forest composition can further slow the spread of a SBW outbreak in the context of the EIS.

AS 12: Multiple management approaches are required for reducing SBW risk in different jurisdictions where different management goals and forest contexts exist (By J.J. Bowden, E.R.D. Moise, R.C. Johns)

Evidence

Key points:

- Forest structure has a significant influence on SBW defoliation, where increases in balsam fir content (and concurrent decreases in hardwood content) result in increased damage.
- SBW outbreaks are fuelled by mass moth dispersal from adjacent regions undergoing an outbreak, with immigrants augmenting local reproductive output and facilitating escape from control by natural enemies (predators that feed upon SBW).
- Even when the EIS is considered a feasible approach, urban spaces, proximity to water bodies, privately owned land and protected natural areas all present logistical challenges to the implementation of SBW management.
- Additional logistical challenges include limited road access to forest stands in many jurisdictions, necessitating sampling by other costlier and more logistically challenging means (e.g., helicopter).
- Newfoundland, as a northern island, is relatively low in biodiversity and, therefore, likely features a lower diversity of natural enemies to attack rising populations of SBW. Community interactions can also be indirect via other co-occurring herbivores (e.g., introduced species) and exacerbate impacts.

Details:

Forest composition has a significant influence on SBW defoliation, where increases in balsam fir content (and concurrent decreases in hardwood content) result in increased damage (Zhang et al. 2018). Moreover, modelling results published by Gray (2013) indicate that both the duration and severity of SBW outbreaks are strongly, positively correlated with balsam fir content. Given that balsam fir is the dominant tree species of western Newfoundland, for example, it is unsurprising that both the previous and current SBW outbreaks on the island originated in this region. Similar stand types in other jurisdictions (e.g., Ontario) have also been implicated in promoting SBW outbreaks (Blais 1985). Although SBW is found throughout much of Canada east of the Rockies, outbreaks in west-central Canada are very different, again most likely because of the spruce-dominated and mixedwood forests in the latter region and the reduced incidence of balsam fir (Brandt and Amirault 1994). Beyond forest composition, SBW outbreaks are fuelled by mass moth dispersal from adjacent regions with outbreaks, with immigrants augmenting local reproductive output and facilitating escape from control by natural enemies (Pureswaran et al. 2016; Johns et al. 2019). Owing to the prevailing westerly winds, the transportation of moths from mainland outbreaks to western Newfoundland has been repeatedly observed in recent years, coinciding with the SBW population increase that began on the island in 2018.

The spread of outbreaks may be further augmented by unmitigated local source populations. Since the beginning of the outbreak in Quebec in 2006 (prior to the existence of a proactive control program focused on containing hotspots), population growth occurred beyond possible mitigation by the EIS, necessitating the implementation of the FPS to minimize damage to high-value stands. However, even when the EIS is considered a feasible approach, urban spaces, proximity to water bodies, privately owned land and protected natural areas all present logistical challenges to the implementation of SBW management. In western Newfoundland, for example, Gros Morne National Park comprises an extensive forest landscape (~800 km^2), while also serving as the epicentre of the current SBW outbreak. Following policy review and public consultation, a decision was made by Parks Canada to decline participation in the EIS and allow the outbreak to proceed as a natural process. Accordingly, the value of such spaces becomes a function of their use as control blocks against which to compare responses observed in treated forest stands. For example, a current collaboration between the Canadian Forest Service and Parks Canada aims to ascertain the downstream consequences of severe defoliation and subsequent tree mortality on forest community structure. From the perspective of the management program, SBW sampling inherent to such research activity contributes to the monitoring of rising populations. Conversely, these unmitigated outbreak hotspots necessitate additional resources to impede outbreak spread into the neighbouring landscape.

Additional logistical challenges include limited road access to forest stands in many jurisdictions. Multiple bouts of branch sampling for second instar larvae are integral to the EIS (Johns et al. 2019), providing data on budburst phenology, treatment efficacy, and budworm population forecasting. Such high density and frequent sampling, however, is only economically and logistically feasible where road networks allow (relatively) convenient travel by vehicle. Although such networks are present in New Brunswick due to extensive forestry activity and a smaller area, resource roads are much more spatially limited in rural areas of Ontario, Quebec, and Newfoundland and Labrador, necessitating sampling by other costlier and more logistically challenging means (e.g., helicopter). This increase in resource allocation and collection time ultimately comes at the cost of reduced sampling density and, owing to the role of interpolation in developing budworm population maps, less certainty and precision in contouring outbreak hotspots.

Lastly, inherent to the EIS is the knowledge of SBW interactions with the broader forest community. The SBW natural enemy community, for example, plays an important role in regaining control of budworm populations following treatment application (Johns et al. 2019). This community can be highly complex, and although it is well characterized for mainland forest ecosystems, including Ontario and Quebec (e.g., Eveleigh et al. 2007; Greyson-Gaito et al. 2021), comparatively little is known about these relationships in other jurisdictions, including Newfoundland and Labrador. Newfoundland, being a northern island, is relatively low in biodiversity and, therefore, likely features a lower diversity of natural enemies to attack rising populations of SBW. However, Bowden et al. (2023) recently showed that spiders frequently consume SBW in rising populations in Newfoundland, and while spiders do not mount a density-dependent numerical response, there is scope for a functional response to rising SBW populations. Community interactions can also be indirect via co-occurring herbivores. The island of Newfoundland has many introduced species (South 1983), some of which are capable of significantly altering the landscape, such as the hyper-abundant moose (McLaren et al. 2004). Furthermore, stands damaged by SBW are not likely to undergo natural succession/regeneration due to herbivory by high moose populations (McLaren et al. 2004). It is yet unclear to what extent these stressors, in combination with SBW defoliation, could amplify the negative effects on the spruce-fir forest.

Ultimately, the circumstances in most jurisdictions outside of New Brunswick are sufficiently different that multiple approaches to SBW management are likely warranted (see also sections on Management, AS 13 and 14, for other management options). As the EIS is a novel approach to management, no long-term data yet exist, and we can therefore not yet predict well how the current outbreak may progress in jurisdictions outside of New Brunswick.

Uncertainty

1. High uncertainty whether some jurisdictions, owing to their size and geographic complexity, might benefit from a hybrid strategic approach using both the EIS and the FPS in tandem to manage a SBW outbreak.
2. High uncertainty regarding the structure and function of the SBW natural enemy complex (e.g., community diversity) in some jurisdictions, particularly in its ability to limit the growth of SBW populations in the context of the EIS.
3. High uncertainty regarding the degree to which SBW populations can be managed under EIS in contexts where long-distance and within-province moth dispersal will interact to influence the temporal and spatial scale of the outbreak.
4. Moderate uncertainty that trees stressed by introduced insect pests could amplify the risk of tree mortality.
5. Moderate uncertainty that jurisdictions will have the capacity (e.g., aircraft and staff) to treat all priority areas if the outbreak area expands significantly.
6. Moderate uncertainly regarding the extent to which scale-dependent, untreated hotspots (e.g., in forest stands that may have different forest management objectives) will impact the success of the EIS within forest landscapes, including across jurisdictions (e.g., federal, provincial, private forests).
7. Moderate uncertainty regarding the extent to which a SBW management program versus no management approach will impact the broader forest ecosystem (i.e., ecosystem services).
8. Moderate uncertainty regarding the extent to which reduced budworm monitoring capability (i.e., less than New Brunswick) impacts the efficacy of the EIS.
9. Low uncertainty that the existence of balsam fir-dominated forests and high propensity of moth immigration events are characteristics that increase susceptibility to outbreaks for other jurisdictions
10. Low uncertainty that limited forest access negatively impacts SBW monitoring capability to implement an EIS.

Reducing uncertainty

1. Assess, through both field and modelling studies, the circumstances under which the EIS might be used in tandem with the FPS to manage a SBW outbreak.
2. Develop tools to reliably detect local and immigrant moths to quantify their combined and individual contributions to rising SBW populations.
3. Determine, through meta-population modelling, how the size, number, and proximity of SBW hotspots influence the spread of the outbreak.
4. Develop tools to overcome logistical constraints of limited forest access that provide data pertinent to the EIS (e.g., predictions of budburst phenology and stand susceptibility to defoliation).
5. Determine non-target and downstream consequences of the EIS versus a “no-management” strategy on forest community structure and function.
6. Determine the structure and function of the SBW natural enemy complex, and its role in and response to the EIS.
7. Assess the extent to which co-occurring biological stressors (e.g., herbivory by moose, invasive insects) amplify the effects of budworm outbreaks on tree mortality and forest resilience.

Management – Foliage Protection Strategy

AS 13: An FPS can reduce SBW impacts but will not significantly alter the timing, duration, or locations of the outbreak (By B.J. Cooke, P. Therrien, L. Morneau)

Evidence

Key points:

- Evidence exists for the continued efficacy of Bt or tebufenozide against SBW, and there is also abundant evidence that SBW larval populations may be reduced to levels where sufficient foliage is protected to keep host trees alive, even during intense outbreaks that last for many years.
- Operationally treated areas are typically small, in the order of tens to hundreds to thousands of hectares. During the last outbreak in Quebec, no more than 4–5% of the province's forests were sprayed for SBW. Although budworm populations are reduced by an FPS in these treated areas, the areas are never large enough to have any substantive impact on the wider SBW population.
- Some outbreaks collapse even when the forest is not heavily damaged, and substantial evidence indicates that rising late larval mortality caused by specific natural enemies, along with microsporidian disease are critical to such collapses. Thus, the greater the role of natural enemies in regulating SBW population cycles, the lesser the role of SBW-induced forest decline, and the less likely that spraying insecticide mid-outbreak prolongs the outbreak.

Details:

The development and use of *Bacillus thuringiensis* (*Bt*) and its variants in Canada are described in the comprehensive review by van Frankenhuyzen (1990). For decades, SBW populations have been treated with a range of aerial insecticides, with varying degrees of efficacy (Prebble 1975; Armstrong and Ives 1995). Today, the most commonly used insecticides contain *Bacillus thuringiensis* var. *kurstaki, "Btk",* an ingestible stomach toxin with two modes of action that have been well described. Insecticides based on tebufenozide are also registered in Canada for use against SBW (see also AS 15).

The efficacy of *Btk* insecticide spraying was a subject of controversy at the tail end of the last outbreak, in the late 1980s, leading to the development of an ecophysiological model of dose acquisition and dose expression (Cooke and Régnière 1996), which was subsequently validated (Régnière and Cooke 1998), proving beyond a doubt that the product was indeed highly efficacious when applied using fixed-wing aircraft. Cooke and Régnière (1999) subsequently showed that predicting efficacy and measuring efficacy in the field requires high-quality data of a standard that is rarely available under most operational situations. Cooke and Régnière (1996) showed that the optimal combination of spray parameters (timing, droplet size, droplet density) would depend on the goal of the program: population reduction or foliage protection. Early sprays (targeting larval stages L3-L4) with fine droplets were predicted to offer better foliage protection, whereas late sprays (targeting larval stages L5-L6) with large droplets were predicted to offer better population reduction. Such insight allows pest managers to maximize the probability of meeting their program objective, be it foliage protection or population reduction, or some intermediate objective.

Evidence has continued to mount in favour of the continued efficacy of *Bt* in use against SBW: in eastern Quebec in the 1970–80s (Fuentealba et al. 2022), in western Quebec in the 1990s (Bauce et al. 2004), and in Côte-Nord, Quebec, in the 2000s (Fuentealba et al. 2019). In summary, there is abundant evidence that larval populations may be reduced to a point where sufficient foliage is protected to keep trees alive, even during intense outbreaks that last for many years.

These operationally treated areas are typically small, in the order of tens to hundreds to thousands of hectares, but the protective benefit should scale up, limited only by the economics and policies of aerial applications. During the last outbreak in Quebec, no more than 4–5% of the province's forests were sprayed for SBW. This is the spatial scale across which foliage protection benefits are realized. Although budworm populations are temporarily reduced in these treated areas, the areas are never large enough to have any substantive impact on the wider SBW population. The most significant spray programs ever conducted in Canada were in New Brunswick, through the previous outbreak of the 1970–1980s. Here, it was demonstrated that although insecticide spraying reduced defoliation levels in sprayed areas, the duration of the outbreak cycle was identical in sprayed and unsprayed areas (Gray and MacKinnon 2007). This was attributed to moth dispersal and egg recruitment, which tends to homogenize population densities in different areas (Royama et al. 2005).

The effectiveness of insecticide spraying against SBW received considerable scientific scrutiny in the 1970s and 1980s because of early forest-budworm models (reviewed in Sturtevant et al. 2015) that presumed that forest condition was both necessary and sufficient to explain the periodic recurrence pattern of outbreaks. Baskerville (1975) argued that an abundance of overmature fir would increase the duration of outbreaks, while Blais (1974) argued, complementarily, that an abundance of overmature fir would hasten the recurrence of SBW outbreaks. These presumptions led to a consensus view amongst critics of insecticide spray that maintaining the forest in a relatively green state during a SBW outbreak would only prolong the length of outbreak. Absent from these models, and from this policy position, was any role for rising populations of specialist natural enemies (various species of parasitic wasps [Hymenoptera] and flies [Diptera]) in assisting in the termination of a cycle. In retrospect, this was a significant shortcoming, because we now know that some outbreaks collapse even when the forest is not heavily damaged (Royama 1984), and there is substantial evidence that the missing element is rising late larval mortality caused by these specific natural enemies, along with microsporidian disease (Royama et al. 2017). The greater the role of natural enemies in regulating SBW population cycles, the lesser the role of SBW-induced forest decline, and the less likely that spraying insecticide mid-outbreak prolongs the outbreak.

This issue of the importance of natural enemies in helping to precipitate outbreak collapse is addressed in the section on SBW population dynamics and outbreaks (AS 1 and 2).

Uncertainty

1. Medium uncertainty that natural enemies are capable of terminating outbreak cycles before there is widespread stand mortality, particularly in spruce stands (as discussed in the section on Population dynamics and outbreaks, AS 1 and 2).
2. Low overall uncertainty for the affirmative statement. Consensus in the research community is strong.

Reducing uncertainty

1. Use and refine simulation models to explore the consequences of various assumptions about the role of natural enemies in the context of insecticide application, because the role of natural enemies in precipitating collapse in sprayed stands requires increased attention.
2. Study alternative control products for suppression of SBW to diversify the range of available products. *Btk* is the dominant insecticide currently used in aerial applications programs, but not the only insecticide registered for use against spruce budworm. Although there is no evidence of resistance to *Btk* in field populations, and it is unlikely to evolve at the low levels of selection pressure applied in the Canadian forestry context, resistance can be bred in the highly controlled conditions of the laboratory.

Management – Pre-emptive harvest (redirected and accelerated) and salvage logging

AS 14: Pre-emptive harvest and salvage logging are alternative approaches for SBW management (By B.J. Cooke, P. Therrien, L. Morneau, J.P. Brandt)

Evidence

Key points:

- Integrated pest management considers the full suite of strategies and tactics available for managing pests such as SBW, and pre-emptive harvesting and salvage logging are two viable alternatives.
- There are economic and ecological considerations associated with both pre-emptive harvesting and salvage logging, and these should be weighed carefully to select the most appropriate option and to ensure that sustainable forest management objectives are met.
- From an economic standpoint, pre-emptive harvest and salvage logging can represent an economic opportunity, but also an economic risk, depending on the scale of the outbreak, mill and product considerations, and market conditions.
- From an ecological standpoint, pre-emptive harvest and salvage logging should be conducted to minimize any negative ecological effects for present landscapes, and to ensure that forest management prescriptions followed today do not lead to enhanced SBW susceptibility and vulnerability of future forests when the next outbreak cycle arrives.
- Optimal decision making requires robust data and information regarding all alternatives.

Details:

In an integrated pest management framework, all options available to forest and pest managers must be considered carefully when trying to minimize the negative impacts of an insect outbreak or pathogen epidemic. In the case of SBW, two direct methods of management, the EIS (proactive) and the FPS (reactive) have been discussed thus far (AS 10, 11, 13). Silvicultural approaches afford another set of options for forest managers when dealing with a SBW outbreak. Timber harvests in eastern Canadian provinces are typically scheduled decades in advance of actual harvesting operations. As a SBW outbreak cycle begins to emerge, and then peaks some 5–10 years later, there are opportunities, first, for (i) pre-emptive harvesting, including harvest re-scheduling, and accelerated harvesting of host stands expected to be defoliated and otherwise damaged by SBW, and (ii) salvage logging, which we define as the harvesting of trees that are dead, dying, or deteriorating (i.e., materially damaged by SBW) before the fibre of the affected trees loses its economic value (Forestry Canada 1992; Smith et al. 1997; Ashton and Kelty 2018). Salvage logging is also used in forest stands affected by other damage agents, such as fire, weather, pathogens, or other insects. During a SBW outbreak, defoliation can persist for years, leading to growth loss, crown dieback, and eventually, tree mortality and rot (Basham 1986). It is important to realize that SBW mortality occurs progressively, so that there is a continuous range, between 0% and 100%, of stem mortality expected in salvaged stands. In any salvaged stand, some portion of the stand will contain live trees with varying levels of defoliation. Some authors make a distinction between salvage and pre-salvage, with salvage applying only to stands with mortality and pre-salvage applying to stands likely to be defoliated and die (see Blum and MacLean 1985; Muzika and Liehold 2000; Nyland 2016); we do not, however, make this distinction because the latter is essentially equivalent to pre-emptive harvesting. All silvicultural approaches represent economic and ecological benefits and risks.

The ability to forecast future budworm damage can provide guidance as to which stands to harvest first in the harvest queue. Wood supply analysis is used to predict the consequences of an altered harvest queue. Damage forecasts (or impact scenarios) can provide information to decision makers to allow either raising the annual allowable cut (AAC) in response to anticipated budworm-caused damage and tree mortality, or harvesting above the rate allowed under normal circumstances. The more precise and evidence-based the forecast scenario, the greater the likelihood the modified operations will meet sustainable forest management objectives and be acceptable to forest stakeholders.

Modified harvest scheduling operations can represent an economic opportunity, but also an economic risk. Trees to be harvested are at their most valuable before defoliation occurs, with value declining with dieback and mortality (resulting from defoliation). Dead and dry trees are more costly to sort and mill, and thus less desirable to forest companies (Sewell and Maranada 1978). Also, the value of balsam fir, white spruce, and black spruce differ depending on demand for intended products and individual mill requirements. Such factors can be very different, depending on the region. The declining price with declining wood quality creates a strong economic incentive for temporarily raising harvest rates while SBW populations are still in the rising stage and before defoliation reaches 50%. Although pre-emptive harvesting can create wood supply shortages in the longer term, they can be economical in the short term, as long as (a) milling capacity is sufficient (including, for example, the number of mills, the size of mill yards, and the labour supply) and (b) demand and markets exist for intended products. Non-receptive markets can occur when supply is too high, leading to a commodity price collapse. Waiting too long to remove infested stems and stands will lead to a surge in low-quality budworm-killed wood for which there is low demand. The risk of economic consequences is proportional to the volumes available for market, and thus on the extent of the outbreak, the area being pre-emptively harvested or salvaged, and the types of products produced from the various host species that have been cut.

Profit margins in the forest products industry are tight, and disturbance probabilities uncertain enough, that maintaining profitability via salvage may require optimized harvest scheduling and product development, which involves sophisticated mathematical analyses and computer programming (Savage et al. 2010; Mushakhian et al. 2020).

Ultimately, it is the extent and synchrony of SBW outbreak cycles that lead to pulse disruptions in market prices. The more extensive the outbreak, the greater the potential for market disruption. The periodic nature of outbreaks thus creates a periodic incentive to direct harvesting to remove the most susceptible and most vulnerable trees before they have been exposed to several years of consecutive, severe defoliation. This periodic incentive creates an inevitable tension between the goals of sustainable forest management and the economic principle of supply-and-demand market-price stability. Should a SBW outbreak be extensive and synchronous, it may precipitate greater economic risks associated with pre-emptive harvesting and salvage logging (see section on Impacts of outbreaks, AS 6, for further details on economics).

The first wave of the 2006–2020 SBW outbreak has already come to Côte-Nord, Quebec, and is showing signs of starting to decline. Meanwhile, the outbreak cycle in New Brunswick and Ontario and the rest of Quebec is lagging well behind this first wave. The risk of market consequences for salvage operations is tied directly to the risk of large-scale, synchronized outbreak, which is evaluated more fully elsewhere in this document (see section on Population dynamics and outbreaks, AS 1–3). Because the timber volumes at risk vary significantly between provinces, as do mill capacity and feedstock for those mills, the need for and economics of salvage logging will also vary from province to province. For regions within provinces not harvesting close to the AAC, there will be more flexibility, and some SBW-killed stands can be allowed to transition through post-disturbance succession without management intervention providing the ecological benefits (see section on Impacts of outbreaks, AS 7) of such non-action. To be clear, "no action" is part and parcel of any integrated pest management framework implemented by forest managers, but no action involves its own set of risks, including fire (see section on Interaction with fire, AS 5).

Quebec's framework for optimized salvage under the province's Sustainable Forest Development Act references budworm salvage operations as well as options available for modified harvest planning within the context of "aménagement écosystémique" or ecosystem-based forest management. Additionally, the province employs forecast tools for building scenarios that allow it to plan for SBW impacts in advance. Quebec's forests are extremely variable, so some regions, such as the Bas-Saint-Laurent and Gaspésie are expected to experience larger salvage operations than Abitibi-Témiscamingue, where there are fewer stands at serious risk of budworm-caused damage and mortality. Modified harvest forecasting scenarios can be produced for any region and adjusted as the outbreak progresses.

Ontario's regulatory framework is outlined in the Forest Management Planning Manual, which is regulated under the province's *Crown Forest Sustainability Act*. As SBW impacts become increasingly imminent, a regional planning team develops an integrated pest management response plan that includes options for salvage harvest, redirected harvest, accelerated harvest, prescribed burning, insecticide treatment, and no action. The regional planning team runs iterations of the Sustainable Forest Management Model to predict the impacts on Forest Management Unit wood supply from various levels of SBW impacts and different combinations of treatments.

In New Brunswick, different management scenarios can be modelled using wood supply analysis tools created with Woodstock together with the SBW Decision-

Support System (DSS). The DSS can be applied to examine different effects of modifying harvest schedules on wood supply and evaluate management responses and modifications to the harvest queue (MacLean 1996).

From an ecological perspective, there are risks and positive and negative impacts associated with either outbreaks or any silvicultural approach to SBW management, including no action (see section on Impacts of outbreaks, AS 5, 7). The critical point is that forest management should be focused on ensuring the minimization of risk and impacts, both for the management of the current SBW outbreak and for potential future outbreaks (Burton et al. 2015). Much has been written about the "silvicultural hypothesis" (e.g., Miller and Rusnock 1993; Pureswaran et al. 2016; Robert et al. 2018; Kneeshaw et al. 2021; Cooke et al. 2024) and the potential role that forest management has had and could have on both the susceptibility and vulnerability of future forest stands. Evidence suggests that past forest management practices affect future SBW outbreaks and their impacts (Robert et al. 2012, 2018). Harvesting, whether pre-emptive or salvage, should be conducted in such a way as to minimize any negative ecological effects for present landscapes, and to ensure that forest management prescriptions followed today do not lead to enhanced SBW susceptibility and vulnerability of future forests when the next outbreak cycle arrives. Forest management practices that lead to an increase in the proportion of balsam fir trees outside the natural range of variability for a particular region are likely counterproductive from the perspective of ecosystem integrity and long-term susceptibility and vulnerability to SBW outbreaks.

Paradigms in forest management have continued to evolve as societal needs and goals have changed and scientific knowledge has accumulated (Rotherham and Armson 2016). During the previous two outbreaks of SBW, forest management paradigms shifted from sustained yield management to multiple use management and finally integrated forest resource management. More recently, forest management has focused on broader ecosystem sustainability and resiliency, a fuller range of ecological goods and services, emulation of natural disturbances, and ecosystem-based management. These latest paradigms work towards a broad range of goals, including the maintenance of biodiversity and the integrity and resiliency of ecosystems (in the face of disturbances) by reducing or minimizing gaps between managed and natural forests, while still providing for a variety of goods—including wood—and services (Mathey et al. 2005; McAfee and Malouin 2008; Rist and Moen 2013; Rotherham and Armson 2016; Innes and Tikina 2016). Forests are valued for many reasons beyond timber, with biodiversity, carbon sequestration, and water quality and quantity being critical and current examples, especially in the face of global change. Any management intervention for SBW should factor in these broad goals when deciding on the most appropriate management approach or approaches. Strong examples of the latter in the case of carbon sequestration are those of Gunn et al. (2020) and Liu et al. (2022) (see section on Impacts of outbreaks, AS 7).

The predictive accuracy of the modelled impact scenarios for SBW is limited by the collective understanding of SBW population dynamics and the accuracy of the forest inventory. Research aimed at improving this understanding and reducing uncertainty will thus increase the possibility that modified harvesting will deliver maximum benefits with minimum costs while still achieving sustainable forest management goals.

Uncertainty

1. High uncertainty on the effect of intense, large-scale salvage operations on the SBW natural enemy community and on outbreak risk during the following cycle (see AS 1).
2. Low to moderate uncertainty that sustainable forest management objectives can be achieved under harvest plans that have been modified as part of SBW management.
3. Low to moderate uncertainty that the scale of salvage and modified harvest in eastern Canada will be a fraction of the total outbreak extent.
4. Low uncertainty on the importance of the scale of the modified operation in determining site, stand, and landscape-level risk.

Reducing uncertainty

1. Create better tools for (i) predicting future budworm levels and impacts on multiple values, and (ii) predicting ecological outcomes of proposed interventions.
2. Determine the effect of regulatory constraints on landscape-scale conservation and sustainable forest management goals for areas affected by SBW outbreaks.
3. Enhance understanding of how salvage and pre-emptive harvesting may affect economic and ecological considerations.

Impacts of management on non-target species

AS 15: All SBW insecticidal treatment options have some impact on non-target organisms
(By V. Martel, J.J. Bowden, E.R.D. Moise)

Evidence

Key points:

- Commercially available SBW insecticides (registered under the *Pest Control Products Act*) that are based on either *Btk* or tebufenozide affect lepidopteran larvae that are feeding at the time of applications
- In treatments against SBW, only forests with a high balsam fir and spruce content (i.e., forests most susceptible to SBW) are sprayed, and, therefore, spatial limitations further restrict non-target impacts.
- Recent research has shown no effect on lepidopteran abundance and richness from insecticide treatment (both *Btk* and tebufenozide) applied within the EIS context in New Brunswick the year of treatment or one year later.
- At realistic concentrations of *Btk* and tebufenozide, there is little or no effect on aquatic biodiversity.
- Because parasitized SBW larvae feed less than healthy larvae, and the insecticides must be ingested to be effective, parasitized larvae are less likely to be killed by insecticide treatment.

Details:

Direct impacts

Commercially available SBW insecticides (registered under the *Pest Control Products Act*) that are based on either *Btk* or tebufenozide affect lepidopteran larvae that are feeding at the time of applications. Because there is no commercial insecticide that is specific to SBW, all treatment options are expected to have some impact on non-target organisms. Moreover, this impact is likely to vary by product. Tebufenozide applied to oak trees for management of oak processionary moth (*Thaumetopoea processionea* Linnaeus) affected non-target Lepidoptera (the order of insects that includes all moths and butterflies), and this impact varied by insect family (Leroy et al. 2017). The impact of *Btk* also differs depending on the formulations used, the species targeted, and the instar of non-target species at the time of application (reviewed in Schweitzer 2004). The half-life of *Btk* is 1.6–2 days and thus its impact on non-target caterpillars is relatively restricted in time. In contrast, the half-life of tebufenozide is longer at 20–45 days (Sundaram et al. 1996). Carry-over effects of tebufenozide have been shown in SBW, through reduced mating success and fecundity (Cadogan et al. 2002; van Frankenhuyzen and Régnière 2016), and could thus potentially affect other lepidopteran species too. In treatments against SBW, only forests with a high balsam fir and spruce content (i.e., forests most susceptible to SBW) are sprayed, and, therefore, spatial limitations further restrict non-target impacts (e.g., Nunez-Mir et al. 2021).

Lepidopteran communities, if affected by SBW insecticidal treatments, are likely to recover within a few years through recolonization from neighboring non-treated forests, given the limited spatial scale of applications (i.e., focused on spruce-fir stands of high value for forestry). For example, following a high dose treatment of *Btk* for western spruce budworm, significant declines in lepidopteran larvae were initially observed. Many species began to recover, however, within two years (Boulton et al. 2002; Boulton and Otvos 2004). Similar responses were observed in plots treated with *Btk* against *Lymantria dispar,* where initial reductions in the abundance of most non-target lepidopteran species were reversed by the end of the four-year sampling period, and compositions between treatment and control sites were similar (Boulton et al. 2007). The frequency and recurrence of applications will, however, determine the impacts and legacy effects on non-target species, with yearly applications likely having stronger impacts than treatments applied every two or three years. Under the FPS, where budworm outbreaks proceed, treatment to protect selected stands may be applied in consecutive years, although spraying every second year was deemed the best approach in terms of costs/benefits (Bauce et al. 2018). The newer EIS, which has been employed since 2014 in New Brunswick, has resulted in only a low proportion of sites being treated in two consecutive years (18.9% of hectares treated from 2015 to 2021, D. Carleton, DNRED, personal communication); thus, the likelihood of sustained impacts on non-target lepidopteran species is reduced. A recent study has shown no impact on lepidopteran abundance and richness of insecticide treatment (both *Btk* and tebufenozide) applied within the EIS context in New Brunswick the year of the treatment or one year later (Glaus et al. 2023).

As per insecticide label instructions, water bodies are not directly sprayed during SBW treatments. There is some risk, however, that insecticides may drift to or otherwise accumulate in water. Assessments of these types of risks demonstrated that at realistic concentrations of *Btk* and tebufenozide, there is little or no effect on aquatic biodiversity (see review by Kreutzweiser et al. 2013).

Indirect impacts

SBW treatments might also have indirect impacts on non-target organisms by affecting the availability of hosts or prey for natural enemies, or by altering forest habitat. Any negative impact on lepidopteran larval abundance could negatively affect parasitoids or predators (both invertebrates and vertebrates) that depend on lepidopteran hosts/prey for their development. Immature parasitoids developing in SBW larvae killed by an insecticide would also die. Nonetheless, because parasitized SBW larvae feed less than healthy larvae, and the insecticides must be ingested to be effective, parasitized larvae are less likely to be killed by the treatment (shown for *Btk* by Nealis and van Frankenhuyzen 1990). In addition, the proportion of western spruce budworm larvae parasitized by *Apanteles fumiferanae* and *Glypta fumiferanae* was higher at sites treated with *Btk* compared to control sites (Hamel 1977). These are the two main parasitic species attacking their host as first and second instar larvae and thus developing inside budworm larvae at the time of *Btk* applications. However, Hamel (1977) also showed western spruce budworm parasitism occurring after *Btk* applications was lower in treated sites compared to control sites, likely because hosts had been in contact with insecticides before the parasitoids attacked them. The timing of application and the phenology of the different parasitoid species will thus determine the impacts of SBW insecticidal treatments on parasitoids.

Any reduction in populations of lepidopteran species can also affect predators, such as birds and spiders, through prey availability. As mentioned by Venier and Holmes (2010), the non-target impact of SBW treatments on birds is likely to be different in the FPS, which is used when SBW is abundant and predation intensity is low, compared to the EIS, which is used when SBW densities are low (see section on Impacts of outbreaks, AS 7). In the longer term, tree mortality resulting from an unmanaged SBW outbreak, however, can also have negative impacts on bird populations through a reduction of potential nesting and foraging locations (Holmes et al. 2009), and could also possibly increase nest predation (shown for *Lymantria dispar* in Thurber et al. 1994). In general, despite the potential for insect disturbance to contribute to boreal forest complexity and biodiversity (Bergeron and Fenton 2012), given the extent and severity of SBW outbreaks (Blais 1983; MacLean and Ostaff 1989, but see Zhao et al. 2014 for evidence of spatial patterning of damage), the resultant coarse-scale mortality (particularly in balsam fir-dominated stands, MacLean 1980) suggests that net downstream consequences could be negative. According to Schweitzer (2004), the impacts of *Btk* on vertebrate food source would in some cases be lower than the impact of the defoliation from the target insect species. Similarly, preventing trees from dying by treating against SBW also preserves the forest as a habitat for different species outside of the SBW trophic network, including species at risk like caribou (Labadie et al. 2021). For more information on ecosystem properties and services, see section on Impacts of outbreaks, AS 7.

By decreasing the abundance and density of SBW, insecticidal treatments may benefit other defoliators feeding on the same host trees due to reduced competition. Manderino et al. (2014), for example, concluded that *Btk* application to suppress *Lymantria dispar* defoliation increased Geometrid diversity (specific family within the order Lepidoptera). Such treatments could thus increase the abundance of summer-feeding insect species—not only in the Lepidoptera, but also in sucking insects (Homoptera) or other defoliators such as sawflies (Hymenoptera)—by increasing the amount of foliage available, thus also increasing the diversity of prey and hosts for natural enemies (reviewed in Schweitzer 2004). Not treating a forest insect outbreak might have worse non-target impacts on Lepidoptera than treating with *Btk*; the outbreaking species causes foliage depletion and tree mortality and increases indirect competition, i.e., an increase in the population of the pest's natural enemies will also increase their attack on other species (Scriber 2004). The benefit of insecticidal application further extends to other insect groups. Wayland et al. (2015) suggested that *Btk* could protect beetle species richness by reducing (or preventing) tree defoliation from *Lymantria dispar.*

Uncertainty

1. High uncertainty regarding the indirect impact of SBW insecticidal treatments on summer-feeding insect diversity, on vertebrate species that depend on insects for food, such as birds, and on species at risk.
2. Moderate uncertainty that insecticidal treatments affect the community of parasitoids through a modification of the lepidopteran community.

3. Moderate uncertainty that insecticidal treatments currently employed against SBW have no significant long-term impacts on non-target lepidopteran species feeding at the time of treatments.
4. Moderate uncertainty regarding the recovery time of lepidopteran species affected by SBW insecticidal treatments.
5. Moderate uncertainty regarding the impacts of insecticides in water bodies on non-targets organisms.
6. Low uncertainty on the impact of insecticidal treatments on SBW parasitized at the time of application.
7. Low uncertainty that SBW insecticidal treatments currently employed have a low impact on lepidopteran species not feeding at the same time as SBW treatments.
8. Low uncertainty that the impacts of SBW insecticidal treatments on other lepidopteran species increase with longer half-life of the insecticide used.
9. Low uncertainty that yearly treatment applications will have greater impacts on non-target lepidopteran species than less frequent applications.

Reducing uncertainty

1. Quantify the impacts of SBW insecticidal treatments on other lepidopteran species (spring-, summer- and fall-feeding) depending on the strategy (EIS vs FPS), the treatment product (*Btk* vs tebufenozide), the spatial scale of application, as well as the frequency of treatments.
2. Quantify the time needed for recovery of lepidopteran species affected by SBW insecticidal treatments applied at different spatial (e.g., stand types and sizes) and temporal scales (e.g., timing and frequency of treatments).
3. Quantify the accumulation of insecticides in water bodies in/near the sites treated for SBW and the impacts of these insecticides on aquatic insects.
4. Quantify the diversity and abundance of summer-feeding insects (Lepidoptera and other insects) and parasitoids in treated versus untreated forests of similar composition, age, and structure.
5. Quantify the mortality inflicted by the insecticidal treatments to parasitoids developing in SBW or other non-target species feeding at the same time.
6. Quantify the diversity and abundance of vertebrates, including birds, in sites treated against SBW vs. untreated sites.

Risk Characterization

Our characterization of risk is necessarily at a high level. Geographically, we refer to broad areas in the range of SBW from Ontario east to the Atlantic provinces as opposed to specific forest stands. Additionally, the nature of the risk is described qualitatively and comparatively, rather than quantitatively. Temporally, unless stated otherwise, our risk characterization refers to the next 10–15 years. We characterize risk using the two fundamental elements of pest risk: likelihood of occurrence and consequences of such an occurrence.

Likelihood of occurrence

Likelihood of occurrence here refers to the occurrence of a SBW outbreak, not merely the occurrence of populations of this defoliating insect. We define an outbreak as a buildup in abundance of SBW that occurs in a given area, causing sufficiently high defoliation, for a sufficiently large area that it can be observed during aerial surveys (Gray 2008). In characterizing this element of risk, we use the relevant evidence and uncertainty described in this report and attempt to answer the question, where will outbreaks occur and when, and account for what can be (or is being) done to prevent them or minimize their probability—that is, how will mitigation activities affect where and when outbreaks occur.

Where will outbreaks occur and when?

Spruce budworm outbreaks are currently occurring in many regions of eastern Canada. It is very likely that these outbreaks will continue and increase in extent to other susceptible areas nearby in the next few years (AS 1). In Quebec, it appears that the outbreak has reached its peak. However, it will likely continue to expand to susceptible areas around the current outbreak (e.g., western Quebec), even though there was a recent decline in the total defoliated area in Quebec. The outbreak reaches farther north than earlier ones in the 20th century and before (AS 1). The EIS is not effective against a well-established outbreak like that currently occurring across extensive areas of Quebec (AS 2, AS 10, AS 12). Although it only began a few years ago, the outbreak in eastern Ontario near the Quebec border is intensifying, growing in extent, and may expand to occupy an area similar in size to previous outbreaks in that area. Changes in forest composition toward higher hardwood content and reduced SBW-susceptible spruce-fir content in western Ontario and northwestern Quebec have happened during the last 35 years (Marchais et al. 2022). Outbreaks, although they are still expected to occur, may occupy a smaller area overall than in the past. Figure 6 shows change in the composition of tree species from 1984 to 2019 (adapted from Hermosilla et al. 2022). We have drawn polygons on the map to help identify large geographic areas of significant species change relevant to SBW. Comparison of the two maps reveals significant changes during the past 35 years from SBW-susceptible forests (spruce-fir) to more resistant tree species in both Ontario and Quebec. Temporally, the outbreak in western Ontario will likely lag behind the rest of the province in terms of both outbreak initiation and collapse—by around ten years, based on experience with previous outbreaks.

What can be (or is being) done to prevent them or minimize their probability?

The irruptive nature of SBW outbreaks, which is an important part of the EIS theory, is probably scale dependent, but there is high uncertainty about this dependency and therefore the preventability of outbreaks (AS 2). It is likely that the feasibility of preventability is reduced as the area-wide population cycle rises in the surrounding regions (AS 2). Due to the effectiveness of the EIS thus far (AS 11), we expect that New Brunswick will not reach an outbreak level (but will have pockets of defoliation) if the EIS continues to be applied in the same manner for the next few years. Beyond that time frame, if the EIS is ceased, the likelihood of an outbreak occurrence is currently not known. Immediate cessation of the EIS would likely result in the development of an outbreak in northern New Brunswick, because the population pressure from Quebec is still significant in that area (AS 3). Populations in Cape Breton, Nova Scotia, are beginning to rise. This development lags behind New Brunswick by a few years. If the EIS is applied as in New Brunswick, one would expect a similar low risk as described above. It is possible, however, that repeated long-range dispersal of immigrant moths from Quebec into Cape Breton could increase the risk in Nova Scotia (AS 3). Also, if the EIS were to end in New Brunswick today, the expected outbreak in northern New Brunswick would likely increase population pressure in Nova Scotia in the next few years.

Long-distance dispersal can be an important part of outbreak development and spread (AS 3). The high populations, the large geographic extent of the outbreak, and the prevailing winds in eastern Quebec make it a likely source of long-distance dispersal to more easterly provinces. Northern New Brunswick, western Newfoundland (island) and northeastern Nova Scotia (Cape Breton) are the most likely recipients of these immigrants (AS 3). Early detection and response to mass immigrations is key for an effective EIS, and sustained prevention requires post-migration management (AS 3, AS 10). Populations have been growing quickly on the western side of the island of Newfoundland. Insecticide treatments under an EIS have been applied for several years, but the effect on the development of the outbreak is largely unknown to this point. The controlling influence of natural enemies on SBW in Newfoundland and Labrador is not yet known (AS 12). If it is less than in New Brunswick and other regions of Canada, we would anticipate a greater likelihood of an outbreak on the island of Newfoundland than in other Atlantic Provinces. Prince Edward Island will likely have low populations of SBW (i.e., below the irruptive threshold), and no outbreak is expected.

In areas where an FPS, salvage harvest and/or pre-emptive harvest is being applied, these actions will not change the trajectory of an existing outbreak (AS 12, AS 14). Monitoring currently undertaken by provinces to inform any of these management approaches appears sufficient to meet their objectives (AS 9). If the level of monitoring for hotspot detection in a particular jurisdiction targeted for an EIS is insufficient to detect small but growing populations before they reach an irruption threshold, the EIS may fade as a valid management option, as populations can quickly increase beyond the suppression capability of such a strategy (AS 2, AS 3, AS 9).

Consequences of occurrence

In characterizing risk within this element, we draw on the relevant evidence and uncertainty described in this report and attempt to answer the question: What will outbreaks do to spruce-fir forests and the values we derive from them? How will mitigation activities affect the consequences of outbreaks?

What will SBW outbreaks do to spruce-fir forests?

Loss of commercial volume. All areas in eastern Canada that experience multi-year outbreaks will lose commercial spruce-fir volumes through mortality and growth loss. These losses will affect the fibre supply of nearby mills (AS 6). Vulnerability of individual stands and regions will vary depending on the proportions of various host species, with those containing high numbers of fir most vulnerable (AS 6). Tolerance for this impact will vary from region to region (AS 6). Regions with an underutilized timber supply will be able to absorb such timber losses because other non-damaged or less-damaged areas can be harvested instead (AS 6). Most susceptible areas in Ontario, Nova Scotia, and the island of Newfoundland fall into this category. On the other hand, New Brunswick, which has a fully committed timber supply, cannot absorb any loss of timber volume (due to either growth loss or tree mortality), and unmitigated outbreaks will directly affect the province's forest industry. Quebec falls between these cases in terms of level of wood supply consequences.

Altered fire risk. There is a clear but complex relationship between outbreaks and wildland fire risk (AS 5). Dead balsam fir stands will increase the frequency and/or size of burned areas in most parts of eastern Canada that are susceptible to SBW outbreaks (AS 5). This risk will tend to lag behind the SBW-caused mortality by a few years. The changing climate and associated future weather patterns are expected to increase the complexity of the SBW-fire relationship, making assessment of risk more difficult (AS 5).

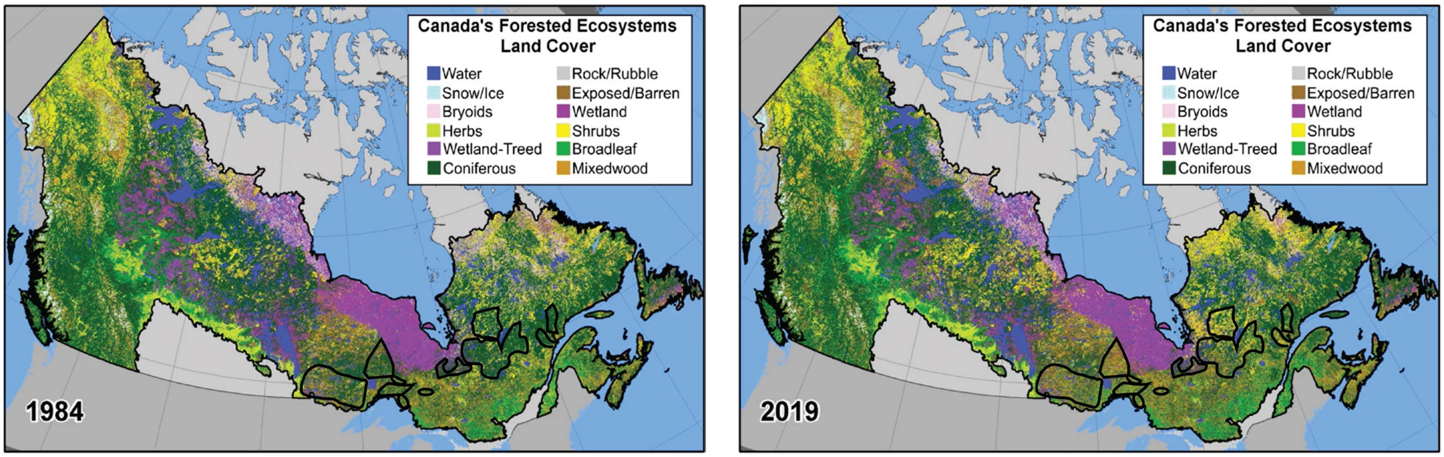

Figure 6. Change in the composition of tree species from 1984 to 2019 (adapted from Hermosilla et al. 2022). Polygons were added to identify large areas of significant species change relevant to spruce budworm.

Impacts on ecosystem dynamics. A reduction in ability to sequester carbon is possible as SBW outbreaks can convert areas from carbon sinks to carbon sources (AS 7). This is more likely in areas not managed for SBW. For areas where SBW outbreaks periodically occur, ecosystem integrity is expected to be resilient to outbreaks in the long term, but short-term negative effects on some ecosystem services (such as immediate successional changes; decrease in NEP and the resulting decrease in carbon sequestration) can be expected (AS 7). In most of eastern Canada, SBW outbreaks are an intrinsic component of forest dynamics, and we therefore expect that affected stands will be resilient across long timescales if such outbreaks occur within the range of natural variability and do not interact negatively with other disturbance agents or factors (AS 7). Immediate successional effects will vary with pre-outbreak stand characteristics and may be pronounced. Novel habitats (not adapted to SBW due to climate change-driven northern range expansion of the insect) are likely to be more susceptible to damaging effects and may also transition to an undesirable new state (AS 4, AS 7, AS 8). Biodiversity changes caused by SBW in most of its traditional range are expected to be within the natural range of variability (AS 7). With the expected shift/expansion of SBW outbreaks to regions farther north, we expect increased impacts, depending on the severity of the outbreak, on cultural services from the forest for Indigenous and rural communities, and in protected areas (AS 4, AS 8). The nature and severity of these impacts are largely unknown but are likely to increase in the long term as the climate continues to change (AS 4). Negative impacts to elements of forest ecosystems (both direct and indirect) from SBW aerial application programs, either in an FPS or an EIS, are possible but unlikely because spray technology is vastly improved with respect to spray delivery, distribution and drift, and the insecticide products used have far fewer unintended effects since previous outbreaks (AS 15).

Other potential impacts. Negative effects on the tourism industry are possible, especially in areas where outbreaks go unimpeded, such as national and provincial parks (AS 8).

How will mitigation activities affect the consequences of outbreaks?

In areas using an FPS for SBW, such as Quebec and Ontario, treated areas (including those where repeated applications may be required) will sustain some growth loss in balsam fir and spruce, but little mortality is expected in these areas during the course of the outbreak, especially in the less susceptible spruce (AS 13). Accordingly, all negative impacts in treated areas will be significantly less than those of an unmitigated outbreak. Salvage and pre-emptive harvesting strategies are expected to continue in Quebec and may begin in Ontario in areas not following an FPS (AS 14). As with all integrated pest management strategies, the intent would be to meet forest management objectives and mitigate negative consequences (AS 10–14). It must be noted that only a small percentage of the defoliated areas in these provinces are treated relative to the total area of the outbreak, but these forests are also the most productive and important for other ecosystem goods and services.

Risks arising from interaction with climate change

As described in the section on climate change (AS 4), the SBW system is sensitive to many climate-related factors, making it likely that a changing climate will exert multiple influences during the coming decades. The spruce-fir-SBW system is likely to change, but these medium- to long-term changes are difficult to predict. The changes will probably affect both the likelihood of occurrence of outbreaks (e.g., outbreaks occurring with greater frequency and in novel, northern forests), and the consequences of such occurrences, such as higher impacts to black spruce forests (AS 4). Reducing the myriad of uncertainties described above by collecting additional baseline data and monitoring changes occurring through time will result in a greater capacity to characterize future risk under climate change (AS 4). Short-term fluctuations in the spruce-fir-SBW system because of climatic change can lead to ecosystem surprise by generating temporary disruptions to equilibrium between spruce-fir forests, SBW and climate, which, taken together, could move these systems to a new state.

Risk levels

We assess risk according to a gradient from low to high and differentiated by region, as warranted. We evaluate the risk levels for each province in eastern Canada in turn (see Fig. 7).

Under the current EIS in New Brunswick, the province is at low risk of a SBW outbreak because the population levels for the insect remain at an endemic level below the irruption threshold. If populations grow to exceed the threshold or there is a pivot to an FPS under an outbreak, risk increases to medium-high, and the outbreak would be expected to move across the province during the next few years. Growth loss and mortality may be greater than in previous outbreaks, as it is unlikely that the province-wide application of an FPS (as was done during the previous outbreak) would occur, due to high costs and potential negative perceptions of the public. Rather, applying an FPS to the highest-priority areas is more likely. A switch to a policy of no treatment for SBW (i.e., allowing an unmitigated outbreak) in the province's forests is highly unlikely and is not assessed here.

Nova Scotia is a partner in the EIS as part of the Healthy Forest Partnership (healthyforestpartnership.ca) and is expected to apply the EIS if populations warrant. If long-distance mass dispersal results in increasing populations in the province, the risk would remain low if the EIS is applied in Nova Scotia as in New Brunswick. If an FPS is applied instead, then the risk increases to medium. Less susceptible forests and a less utilized timber supply in Nova Scotia than in New Brunswick also account for the slightly lower risk there.

The Province of Newfoundland and Labrador is applying an EIS on the island but not in Gros Morne National Park, which lies on the western coast of the island. The efficacy of the EIS here remains unknown, as it is too early in the process to detect management effects. The area of susceptible forest on the island is substantial, so the consequences of high levels of defoliation would likely be major. With significant uncertainty, it is expected that the level of risk in Newfoundland and Labrador is higher than in New Brunswick and Nova Scotia but lower than Ontario and Quebec, especially if populations in Quebec continue to decline.

Prince Edward Island is also a partner in the EIS as part of the Healthy Forest Partnership. This province may receive moths through long-range dispersal from Quebec or New Brunswick, but it has a very low risk of a SBW outbreak due to the relatively small quantity of susceptible spruce-fir forests on the island. Thus, low populations of the pest, along with the success currently being achieved by the EIS in New Brunswick, is expected to keep population pressure on PEI quite low.

Quebec has extensive areas of susceptible forests and has been experiencing a widespread outbreak since 2006 across much of eastern and western parts of the province. Although the outbreak appears to have peaked in eastern regions of Quebec and may begin a declining phase back to endemic population levels, several more years of significant defoliation are likely (i.e., western Quebec), making the risk high in the traditional SBW-affected areas and medium in the north, where timber losses will be of less commercial significance, but potential ecological impacts may be greater.

Ontario also has extensive areas of susceptible spruce-fir forests. The outbreak areas in northeastern Ontario and in western Quebec are parts of the same outbreak, spanning the provincial border. The cycling of the population in western Quebec suggests that northeastern Ontario will be at medium-high risk for several more years. Northwestern Ontario has a medium risk that may nonetheless be lower than the last outbreak's due to a change in forest composition during the last several decades. This will result in reduced outbreak intensity along with a time lag of ten or more years compared to northeastern Ontario. Population pressure will be internal because the area of likely hotspots is extensive. Additionally, the current outbreak in Minnesota could provide a potential source of immigrant females. In southern Ontario, because of the more recent outbreak in this region, we do not expect another outbreak in the next 10–15 years.

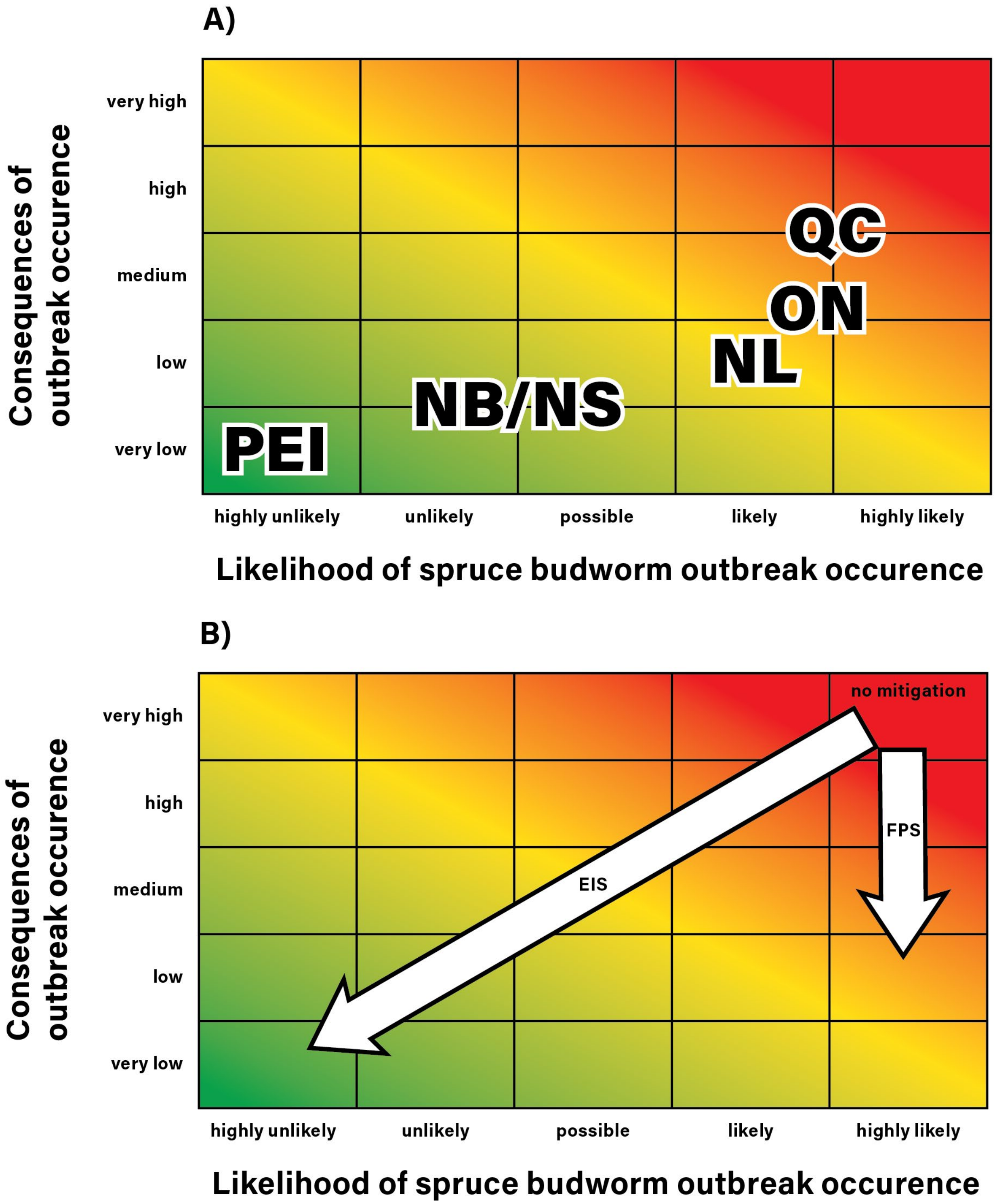

Figure 7. Risk matrix showing a) the level of risk for each province during the next 10–15 years given the expected SBW management strategy/actions, and b) the relative effect of the two primary SBW management strategies on risk level.

Conclusions

Since the beginning of the 20th century, there have been four major SBW outbreaks and, in the first quarter of the 21st century, eastern Canada is experiencing the fifth. During the past 125 years, SBW has had a major impact on the practice of forestry; it is expected to do the same moving into the future as long as the insect continues to negatively affect forest management objectives. Management of SBW has evolved as our knowledge has increased (Alfaro and Langor 2016). For much of the 20th century, the focus was on chemical control, but beginning in the 1970s, emphasis shifted to spraying smaller and smaller areas, moving to biological insecticides, and using other approaches based on silviculture. Pest or forest managers dealing with SBW outbreaks currently use a wide array of tactics. These tactics depend on the forest or land management objectives of a given area. There is also a recognition that there are both costs and benefits associated with SBW and its management. Thus, these need to be weighed carefully when making decisions on the most suitable management option, including no action, for an area.

Paradigms in forest management have also evolved as societal needs and values have changed and scientific knowledge has accumulated (Rotherham and Armson 2016; Alfaro and Langor 2016). During the previous two outbreaks of SBW, forest management paradigms shifted from sustained yield management to multiple use management and, finally, to integrated forest resource management. More recently, forest management has focused on broader ecosystem sustainability and resiliency, a fuller range of ecological goods and services, emulation of natural disturbances, and ecosystem-based management. These latest paradigms work towards a broad range of goals, including the maintenance of biodiversity and the integrity and resiliency of ecosystems (in the face of disturbances) by reducing or minimizing gaps between managed and natural forests, while still providing for a variety of goods, including wood, and non-market services (Mathey et al. 2005; McAfee and Malouin 2008; Rist and Moen 2013; Rotherham and Armson 2016; Innes and Tikina 2016). Forests are valued for many reasons beyond timber, with biodiversity, carbon sequestration, and water quality and quantity being critical and current examples, especially in the context of global change. These spaces also hold cultural value to Indigenous peoples who hold knowledge about these ecosystems as well as use these forests for multiple purposes. Any management intervention for SBW should factor in these broad goals when deciding on the most appropriate management approach or approaches.

Ecosystem management implies taking into account the interactions among components of the ecosystem and how management actions might affect them. A main principle is understanding that SBW is a natural component of spruce-fir forests. This is particularly true for the boreal and hemiboreal spruce-fir forests, where periodic SBW outbreaks are a major part of the large-scale disturbance patterns, along with fire and windstorms. These events also affect each other, adding to, leading to, or inhibiting each other. Indirect management actions (e.g., decreases or increases in balsam fir content following harvesting or wildfire suppression) can exacerbate or dampen these disturbances. Similarly, direct management, such as an FPS during SBW outbreaks or outbreak prevention through EIS, could affect ecosystem values, depending at what scale actions are taken. For an FPS, an important consideration is that although, in general, only a small area of forest is usually treated, this forest is generally the most productive and contains the most high-value timber. In the case of EIS, the SBW population pulse and its consequences are averted. Therefore, SBW management policy should consider the ecological risks of both managing and not managing SBW outbreaks.

During our risk analysis process, we identified important knowledge gaps that need to be filled to reduce uncertainties associated with SBW risks and knowledge. Table 2 provides a summary of 15 of the most important knowledge gaps. The affirmative statements and their associated knowledge gaps can be used to identify priority areas for research. Rising to the top is the recognition that understanding population dynamics is critical to developing appropriate SBW management policies and practices. In particular, it is critical that we better understand what mechanisms suppress and maintain populations at endemic levels, what factors or changes lead to irruptive populations, and what causes the collapse of outbreak populations. These insights are essential for predicting the timing, location, severity, duration, and impacts of SBW outbreaks. It is also essential when selecting management options: EIS, FPS, salvage harvesting, pre-emptive harvesting, accelerated harvest, or no action.

Related to the previous mechanisms is the role of SBW moth dispersal in outbreaks. We need to have a clearer picture of whether outbreaks irrupt on their own across the landscape, or whether moth immigration seeds, accelerates, or synchronizes local population increases that coalesce into large regional outbreaks. We also need to know at what scale these interactions are occurring. Such understanding gives greater confidence in selecting appropriate management strategies.

Indeed, a better understanding of how the interactions in the SBW system affect the health and resiliency of spruce-fir forests will lead to better forest management policies and practices. These interactions include the natural components of the system, such as wildfire, climate, birds, decomposers, small and large mammals, aquatic systems, species-at-risk, soils, and carbon sequestration (i.e., ecosystem goods and services). This understanding must also consider the effects of the management tactics on the overall health and resiliency of forest ecosystems, and the continuum of spatial and temporal scales, from individual stands to landscapes, and short- to long-term, at which SBW management may impact different components of ecological functioning.

Spruce budworm management and broader forest management will benefit from a more comprehensive understanding of the costs and benefits of the available options. The costs and benefits to be considered include economic, social, and ecological. Critically, this also includes the costs and benefits of taking no action. Knowing the relative effectiveness and consequences of the various SBW management options is critical in deciding upon which option, or combination of options, are most likely to provide the most favourable cost-benefit ratio and the desired outcomes. One overriding theme in this risk assessment is the comparison between an EIS and an FPS. Other options, such as salvage harvesting or accelerated harvest, should also be considered. Resource managers need to be able to make informed decisions prior to outbreaks, or at least in the early stages of an outbreak, on which tactics to deploy, when, where, for how long, with what resources, at what cost, and at what scale.

A final and still significant research need is the implications of climate change. Its inherently high uncertainty has the potential to affect all the other questions raised in this risk assessment. It could also disrupt and alter the SBW system so much so that our current understanding of SBW ecology and management is no longer accurate or applicable. Research towards closing SBW knowledge gaps is critical to improving forest management in the face of this insect's outbreaks and minimizing future negative consequences while maintaining long-term forest resiliency.

Table 2. Many uncertainties were identified during the risk assessment process related to SBW outbreaks. The following itemizes elements that need to be addressed to reduce uncertainties related to affirmative statements discussed and reviewed as part of that process, ranked in order of priority. Additional elements requiring further research are listed for each individual affirmative statement.

Affirmative statement	Reducing uncertainty	Authorities identifying the knowledge gap
AS 1: SBW outbreaks are periodic in occurrence and the mechanisms of population irruption are well understood	Determine the mechanisms of population irruption for endemic populations.	B.J. Cooke, J. Régnière, J.-N. Candau
AS 2: SBW population irruptions are preventable	Characterize the factors affecting the transition from focal epicenters to large-scale outbreak using synthetic spatial modeling and validation. So much of our "understanding" comes from theoretical ecology, as opposed to field measurement. This is an urgent area of research if we are to predict budworm risks in the 10–15-year time horizon.	B.J. Cooke, J. Régnière, J.-N. Candau
AS 11: Sustained application of the EIS for the short-term (~5 more years) will reduce the long-term impact of SBW in Atlantic Canada	Monitor, long-term, and evaluate the efficacy of the EIS in Atlantic Canada.	J.J. Bowden, E.R.D. Moise, R.C. Johns
AS 3: Long-distance dispersal of SBW moths is a major factor in outbreaks	Initiate ecophysiological research into factors governing take-off and capacity to sustain flight to improve forecasting the early part of the flight. Similar need exists for wing-folding and descent, followed by host-choice behaviour, particularly for night flight and when moths are forced down by convective storm fronts.	B.J. Cooke, J. Régnière, J.-N. Candau

Affirmative statement	Reducing uncertainty	Authorities identifying the knowledge gap
AS 13: An FPS can reduce SBW impacts but will not significantly alter the timing, duration, or locations of the outbreak	Use and refine simulation models to explore the consequences of various assumptions about the role of natural enemies in the context of insecticide application because the role of natural enemies in precipitating collapse in sprayed stands requires increased attention.	B.J. Cooke, P. Therrien, L. Morneau
AS 10: There are essential elements and principles of the EIS that must be included for a successful approach to managing SBW	Examine the application of EIS principles across a range of jurisdictions, locations, and local outbreak intensities.	R.C. Johns, J.J. Bowden, E.R.D. Moise
AS 4: Climate change is altering the distribution, scale, and intensity of SBW outbreaks in the forests of eastern Canada	Quantify the impact of temperature (means, variability) on SBW's survival, physiology, and life history, particularly at range margins under future climatic conditions.	A.D. Roe, D.S. Pureswaran, J.J. Bowden, J. Régnière, E.R.D. Moise, V. Martel
AS 7: Ecosystem integrity is generally resilient to natural SBW outbreaks but will alter ecosystem properties at shorter temporal and spatial scales	Quantify the long-term impacts of SBW outbreak suppression on forest ecosystems that have evolved under a SBW disturbance regime in the context of climate change	L.A. Venier, M. Stastny, C.B. Edge, and J.J. Bowden
AS 6: SBW outbreaks can result in negative impacts on market and non-market forest values	Improve knowledge on attitudes, preferences, willingness-to-pay, and costs of SBW management alternatives. This should help identify and justify SBW management programs and engage the public, directly affected stakeholders, and Indigenous peoples on options, opportunities, and trade-offs.	D.W. McKenney, E.S. Hope, V. Lantz
AS 14: Pre-emptive harvest and salvage logging are alternative approaches for SBW management	Create better tools for (i) predicting future budworm levels and impacts on multiple values, and (ii) predicting ecological outcomes of proposed interventions.	B.J. Cooke, P. Therrien, L. Morneau, J.P. Brandt
AS 12: Multiple management approaches are required for reducing SBW risk in different jurisdictions where different management goals and forest contexts exist	Determine non-target and downstream consequences of the EIS versus a "no-management" strategy on forest community structure and function.	J.J. Bowden, E.R.D. Moise, R.C. Johns
AS 15: All SBW insecticidal treatment options have some impact on non-target organisms	Quantify the impacts of SBW insecticidal treatments on other lepidopteran species (spring-, summer-, and fall-feeding) depending on the strategy (EIS vs FPS), the treatment product (Btk vs tebufenozide), the spatial scale of application, as well as the frequency of treatments.	V. Martel, J.J. Bowden, E.R.D. Moise
AS 5: SBW outbreaks increase risk of wildland fire	Improve characterization of SBW fuel types and integrate these fuel types into operational tools and fuel maps through empirical research and monitoring.	L.M. Johnston, Y. Boulanger
AS 8: SBW outbreaks will significantly affect cultural services of critical importance to rural and Indigenous communities in managed and protected areas	Improve understanding of cultural services affected – or facilitated – by SBW outbreaks or their prevention, especially in rural and remote communities.	J.J. Bowden, D. Churchill, C.C. Sponarski, M. Stastny
AS 9: Current monitoring methods and intensities are adequate for effective management of SBW	Enhance understanding of what types or combinations of surveillance methods might be used to monitor remote locations in the context of both EIS and FPS (i.e., areas with no road access).	R.C. Johns, D. Carleton, J.P. Brandt

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