ANNEX 3.5

A3.5. Methodology for the Land Use, Land-Use Change and Forestry Sector

The Land Use, Land-Use Change and Forestry (LULUCF) sector of the inventory includes estimates of greenhouse gas (GHG) emissions and removals associated with managed lands and with the conversion of land from one category to another.

As in Chapter 6, the structure of this annex attempts to maintain the land-based reporting categories, while grouping together related data collection and estimate development methodologies. Section A3.5.1 summarizes the spatial framework for estimate development and area reconciliation. The general approach for estimating carbon (C) stock changes, emissions and removals in all forest-related categories, including Forest Land, Forest Land converted to other land uses and Land Converted to Forest Land, is briefly described in section A3.5.2; this description is not repeated for under the Forest Land Converted to Cropland, Forest Land Converted to Wetlands and Forest Land Converted to Settlements subcategories. Section A3.5.3 describes the approach for estimating emissions associated with the use and disposal of harvested wood products (HWP) from wood harvested in Canada and section A3.5.4 describes the methods used to quantify the effect of management practices on agricultural land for the Cropland category. Likewise, the sections on the Grassland (A3.5.5), Wetlands (A3.5.6) and Settlements (A3.5.7) categories focus on category-specific estimation methodologies.

A3.5.1. Spatial Framework for LULUCF Estimate Development and Area Reconciliation

Canada's monitoring system for LULUCF draws on the close collaboration among several scientists and experts in different disciplines. Early on, it was recognized that the approaches, methods, tools and data that are available and most suitable for monitoring human activities in one land category are not always appropriate for another. Differences exist in the spatial framework specific to each land category, and these differences create a risk that activity data and estimates would be spatially inconsistent. A hierarchical spatial framework was agreed upon by all partners contributing to the LULUCF sector to ensure the highest possible consistency and spatial integrity of inventory estimates.

The LULUCF sector of the GHG inventory reports information in 18 reporting zones (Chapter 6, Figure 6–1). These reporting zones are essentially the same as the ecozones of the National Ecological Framework, a hierarchical, spatially consistent national ecosystem classification system (Marshall et al., 1999). For the purpose of reporting LULUCF estimates, three ecozones are split into smaller land units: the Boreal Shield and Taiga Shield ecozones are split into their east and west components to form four reporting zones, and the Prairies ecozone is divided into a semi-arid and a subhumid component (McGovern, 2014). These subdivisions do not alter the hierarchical nature of the spatial framework.

Analysis units are the finest level of spatial resolution and are specific to each estimation system. In managed forests, the analysis units are the geographic intersection of reporting zones (Chapter 6, Figure 6–1) and provincial/territorial forest management units. For the purpose of this assessment, managed forests were classified into 607 analysis units across 12 provinces and territories; Nunavut was excluded because there is no managed forest area in this northern region

Table A3.5–1 **Number of Spatial Analysis Units of Managed Forests per Province and Territory**

Province/Territory	Number of Analysis Units
Newfoundland and Labrador (NL)	24
Prince Edward Island (PE)	1
Nova Scotia (NS)	1
New Brunswick (NB)	1
Quebec (QC)	129
Ontario (ON)	52
Manitoba (MB)	70
Saskatchewan (SK)	40
Alberta (AB)	181
British Columbia (BC)	65
Yukon (YT)	13
Northwest Territories (NT)	30
Nunavut (NU)	0
Canada	**607**

(Table A3.5–1). Changes in the number of spatial analysis units may occur from one submission to the next and reflect refinements in the integration of multiple spatial layers. For example, the modification of administrative boundaries, timber areas and parks can result in units that do not meet the criteria for separate analysis; these units are therefore regrouped.

The most suitable spatial framework for monitoring cropland GHG is provided by the polygons of the Soil Landscapes of Canada[1] (SLC). A soil landscape describes a group of soils and their associated landscapes and provides information, such as surface form, slope, typical soil C content under native and dominant agricultural land use, and water table depth. Soil landscapes are spatially associated with SLC polygons (the analysis units) that may contain one or more distinct soil landscape components. SLC polygons are also the basic units of Canada's National Ecological Framework, a hierarchical, spatially consistent national classification system within which ecosystems of various scales can be described, monitored and reported on (Marshall et al., 1999). The 12 353 SLC polygons are nested in the next level of generalization (1027 ecodistricts), which are further grouped into 194 ecoregions and 15 ecozones. SLC polygons span in the order of 1000 to 1 000 000 hectares (ha) and are appropriate for mapping at the scale of 1:1 million.

Analysis units for estimating the areas of forest converted to other land uses are the result of the spatial intersection of forest conversion strata (Figure A3.5–6) with ecological and administrative boundaries. Forest conversion strata were developed on the basis of expected conversion rates and characteristics. The sampling approach used to monitor forest conversion requires analysis units to be as consistent as possible with respect to the patterns of forest conversion and large enough to provide an acceptable sample size, given the predetermined sampling rate.

The analysis units of different land-use categories can overlap. Most often, the exact location of events within a unit is not known. Therefore, the activity data pertaining to different land-use categories cannot be harmonized at the level of analysis units. The spatial harmonization is conducted within 60 reconciliation units (RUs), which are derived from the spatial intersection of reporting zones with provincial and territorial boundaries. Quality control and quality assurance procedures are conducted at the level of analysis units during estimate development and at the level of RUs during estimate compilation.

The total land mass of Canada reported in CRF table 4.1 is estimated using geomatics layers based on the SLC polygons and quality checked using data from the National Atlas,[2] and includes land area (estimated at 906 676.55 kha) and the area covered by fresh water (estimated at 89 680.51 kha). Estimates of managed and unmanaged forest areas in Canada are based on estimation procedures from Canada's National Forest Inventory (NFI) reconciled with forest areas provided by other data sources described in A3.5.2.5. Reconciliation of total forest area estimates is required due to differences in canopy closure definitions and methods used to develop estimates used in the National Forest Carbon Monitoring, Accounting and Reporting System (NFCMARS) modelling framework and methods used by the NFI. Total national area figures vary for Canada depending on the data source. However, since the SLC polygons serve as the base layer for the National Ecological Framework for Canada, this data source is considered the most appropriate to ensure the consistency of LULUCF areas.

A3.5.2. Forest Land and Forest-Related Land-Use Change

A3.5.2.1. Carbon Modelling

The estimation of C stock changes, emissions from and removals by managed forests, forest conversion to other land uses and land converted to forest land is conducted with version 3 of the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) (Kurz et al., 2009), the most recent of a family of models whose development dates back to the late 1980s (Kurz et al., 1992). The model integrates forest inventory information (stand age, area and species composition), curves of merchantable volume over age, equations to convert stand merchantable volume into total biomass, data on natural and anthropogenic disturbances, and simulations of C transfers between pools and exchanges with the atmosphere that are associated with ecosystem processes and various events.

The ecosystem processes modelled by the CBM-CFS3 to generate the estimates submitted in this report are growth, litterfall, non-disturbance tree mortality and decomposition. The CBM-CFS3 also models events, such as management activities, forest conversion and natural disturbances. Management activities represented are clear-cut, shelterwood harvest, seed tree harvest, selection harvest, commercial thinning, precommercial thinning, salvage logging, residential firewood harvest and the burning of harvest residues. Different practices of forest conversion are also simulated, including controlled burning.

The forest C pools represented in the CBM-CFS3 can be matched with the Intergovernmental Panel on Climate Change (IPCC) forest C pools (Table A3.5–2). Although not shown here, living biomass pools are further subdivided into two sets, for each of hardwood and softwood tree species.

Annual ecosystem process events are simulated as C transfers between C pools executed at each time step (annually) in every inventory record (Figure A3.5–1). During annual processes, C is taken up in the biomass pool and some biomass C is transferred to dead organic matter (DOM) pools. The decay of DOM results in C transfer to another DOM pool (e.g., stem

1 Available online at: http://sis.agr.gc.ca/cansis.

2 Available online at: https://www.nrcan.gc.ca/maps-tools-and-publications/maps/atlas-canada/10784.

snags to medium deadwood pool), to a slow soil pool or to the atmosphere. More information on pool structure and decay rates is provided in Kurz et al. (2009). Rates of C transfer are defined for each pool, based on pool-specific turnover rates (for biomass pools) or decay rates (DOM and soil pools). Turnover rates can be either very high (e.g., 95% for hardwood foliage) or very low (e.g., < 1% for stemwood). Annual decay rates are defined for a reference mean annual temperature of 10°C and exhibit temperature sensitivity according to defined Q10 relationships; the decay rates vary between 50% (very fast DOM pools, such as dead fine roots) and 0.0032% (slow soil pool).

Table A3.5–2 **Forest Carbon Pools in IPCC Guidelines and CBM-CFS3 Modelling**

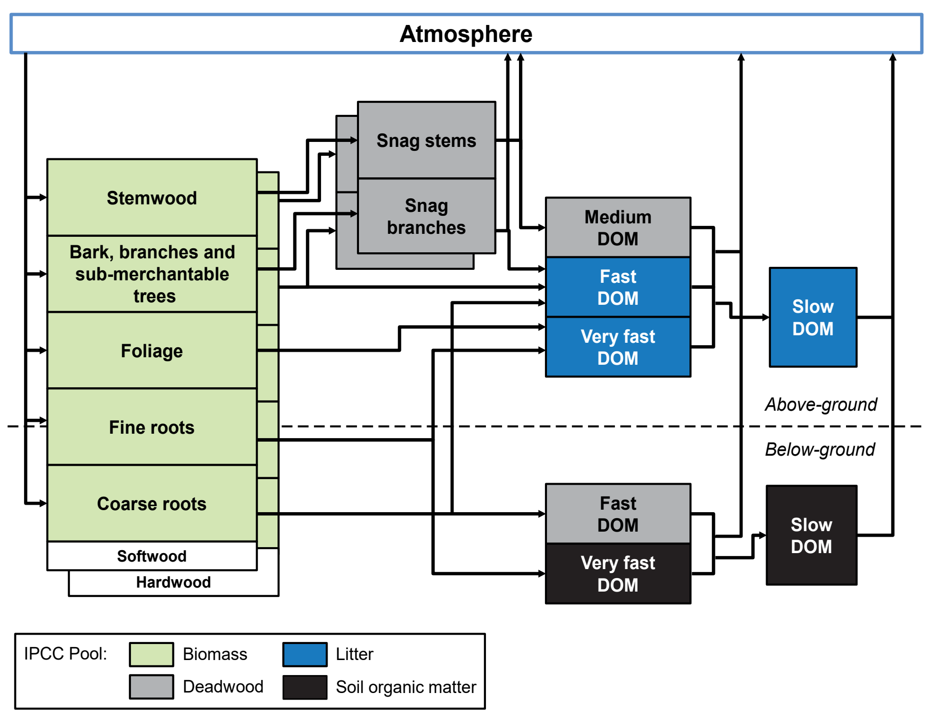
IPCC Carbon Pools		Pool Names in CBM-CFS3
Living biomass	Above-ground biomass	Merchantable stemwood Other (sub-merchantable stemwood, tops, branches, stumps, non-merchantable trees) Foliage
	Below-ground biomass	Fine roots Coarse roots
Dead organic matter (DOM)	Deadwood	Above-ground fast Below-ground fast Medium Softwood stem snag Softwood branch snag Hardwood stem snag Hardwood branch snag
	Litter	Above-ground very fast Above-ground slow
Soils	Soil organic matter	Below-ground very fast[a] Below-ground slow Black carbon[b] Peat[b]

Notes:

a. The below-ground very fast pool includes dead and decaying fine roots, which in practice cannot be separated from soil.

b. Black carbon and peat are currently not estimated.

Figure A3.5–1 **Carbon Pools and Transfers Simulated by CBM-CFS3**



Note: Source: White et al. (2008)

Growth is simulated as an annual process. Each of the records (roughly 3 million) in the 607 analysis units of the forest inventory is associated with a yield curve that defines the dynamics of gross merchantable volume over time. Assignment of an inventory record to the appropriate curve is based on a classifier set that includes province, ecological stratum, leading species, site productivity class and several other classifiers that differ between provinces and territories. Curve libraries for each province and territory in Canada are similar to those used by resource management agencies in the forest planning processes and are derived from permanent or temporary sample plots or from forest inventory information.

Conversion of gross merchantable volume curves to above-ground biomass curves is performed with a set of equations developed for Canada's National Forest Inventory (Boudewyn et al., 2007). These equations derive the above-ground biomass of each stand component from merchantable stemwood volume (per ha), for each province/territory, ecozone, leading species or forest type. Finally, below-ground biomass pools are estimated using regression equations (Li et al., 2003). Mean annual increments are not used in this derivation.

Modelling of C transfers triggered by disturbances is based on the disturbance type and severity, the forest ecosystem affected and the ecological region. For modelling purposes, different practices of forest conversion are also implemented as disturbances. The impact of a disturbance is represented by a disturbance matrix, which specifies, for one or more disturbance types, the proportion of C in each ecosystem pool that is transferred to other pools, released to the atmosphere or transferred to the HWP pool (Figure A6.5–1). In the current submission, the simulation uses a total of 204 disturbance matrices. The number of different disturbance matrices is dependent on the availability of activity data (e.g., the spatial and temporal resolution of disturbance data) and on the knowledge required to parameterize the matrices for more distinct regions or intensities of disturbance.

Within disturbed lands, the amount of C emitted as CO_2 from each pool at the time of disturbance, documented in each disturbance matrix, can be specific to the pool, the types of forest and disturbance intensity, and the ecological zone. There are therefore no CO_2 emission factors applicable to all disturbances of a given type, such as fires. With a few exceptions, the proportion of total C emitted in each C-containing GHG (CO_2, CO and CH_4) due to fire is constant: 90% of C is emitted as CO_2, 9% as CO and 1% as CH_4 (Cofer et al., 1998; Kasischke and Bruhwiler, 2003).

Carbon emissions emitted as CO oxidize in the atmosphere resulting in indirect CO_2 emissions. Amounts of C emitted as CO and indirect CO_2 are calculated by multiplying total C by, respectively, 28/12 and 44/12. More details on the reporting of these indirect CO_2 emissions can be found in Chapter 6 and Annex 7.

While the CBM-CFS3 can model C fluxes at various spatial scales, generating national estimates involves harmonizing, integrating and ingesting vast quantities of data from a large variety of sources. Section A3.5.2.5 documents the key data sources used for this submission.

A3.5.2.2. Post-wildfire regeneration

In previous submissions, a default assumption of 'full regeneration' following wildfire had been used in which all burned areas immediately start regrowing from age 0 on the pre-disturbance yield curve. Recent analysis using remote sensing techniques has demonstrated that regeneration of portions of the forest undergoing severe disturbance may be delayed (White et al., 2017; White et al., 2022). In this submission, a first approximation of average regional (RU-level) regeneration delays was developed and applied to 25% of the area burned by wildfires in each year, based on best available national scale remote-sensing derived information (Hafer et al., 2022; White et al., 2017; White et al., 2022). This approach provides a first estimate of regeneration delay and the impacts of the delay on net flux in the managed forest.

A3.5.2.3. Reforestation

The representation of reforestation activities in the GHG inventory uses certain approaches consistent with afforestation activities (see A3.5.2.8) and others that are very different. A key difference is that "reforestation" occurs on land base that was previously forest and thus must be incorporated into the existing NFCMARS project infrastructure to ensure alignment with provincial/territorial forest inventories and the prevention of complicated double-counting situations.

The basic premise of reforestation is that it takes place on forested land that is not naturally regenerating in an optimal manner (e.g. delayed re-establishment, low density and poor tree resilience). The majority of reforestation in 2021 was targeted for forest that had been burned by wildfire and were not expected to recover naturally due to the severe nature of certain fire events and associated impacts to seed stock and soil quality. A simplifying assumption was established that all reforestation events happen on previously burned forest that is in a state of regeneration delay (see A3.5.2.2). The post-fire delayed regeneration state was considered an acceptable first approximation proxy for forest not recovering naturally. Based on site preparation records of reforestation sites, events were assigned to a residue treatment category and a soil preparation intensity level and 17 new disturbance types and associated matrices were created to represent the range of possible site preparation combinations. More details can be found in Hafer et al. (2022).

A3.5.2.4. Forest drainage

Forest drainage is used to lower the water table, thereby improving soil aeration and promoting root development and tree growth on low-productivity organic soils. A consultation with forestry industry experts and an extensive literature review carried out in 2015 and 2016 suggested that the only province in Canada where operational drainage of organic soils for forestry occurred was Quebec (Gillies, 2016). This management activity occurred from the 1980s through to the mid-2010s on a small percentage of peatlands corresponding to three RUs (11, 12 and 15) on both private and public lands. Forest drainage has progressively declined since 2003 due to the end of government subsidies and changes to Quebec's forest management tenure.

Data on forest drainage were compiled from a combination of historical documents, expert consultations and provincial statistics to develop a time series from 1980–2018 of annual peatland areas drained for forestry on both private and publicly owned forests of Quebec. Provincial statistics (Gouvernement du Québec, 2018) were reported by administrative region (AR) for 1994–2008 and by province for 1986–1993 and for 2009–2017. Drainage data for 1980–1985 were assumed to be constant, resulting in a cumulative area drained equivalent to the 1986 value reported by Quebec statistics, which was also consistent with values cited in Hillman (1987). Given the absence of drainage activity data for 2018 (Gouvernement du Québec, 2018) and the fact that there were no areas drained in 2016 and 2017, drained areas were assumed to be zero after 2017. Estimates of drained areas by AR (1994–2008) were allocated to the three RUs by overlaying the AR to create a spatially weighted area average that was applied to the provincial values for all years.

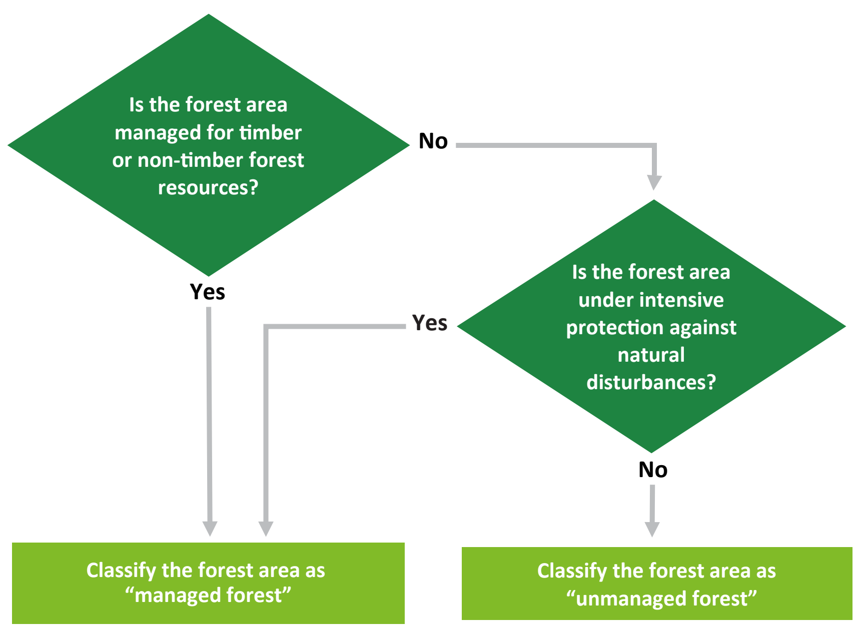
Emissions of CO_2, CH_4 and N_2O from drained organic soils were calculated using a Tier 1 method and emission factors from Tables 2.1, 2.2 and 2.3, respectively, of the 2013 Wetland Supplement to the 2006 IPCC Guidelines (IPCC, 2014). Emission factors are associated with the temperate (RUs 11 and 12) and boreal (RU 15) climate zones. The fraction of area covered by ditches was also determined using the default values for drainage ditches from Table 2.3 of the 2013 Wetland Supplement (IPCC, 2014).

A3.5.2.5. Data Sources

Managed Forest Land

Canada's forests are classified as "managed" or "unmanaged" based on the occurrence of management activities for timber or non-timber and on the level of protection against disturbances (Figure A3.5–2). Managed forests occur within all provinces and territories of Canada, with the exception of Nunavut (Figure A3.5–3). The estimation of the managed forest area required the spatial delineation and combination of boundaries of many different forest areas, including all operational forest management units, timber supply areas, tree farm licences, industrial freehold timberland, private woodlots and

Figure A3.5–2 **Decision Tree for the Determination of Managed Forest Area**



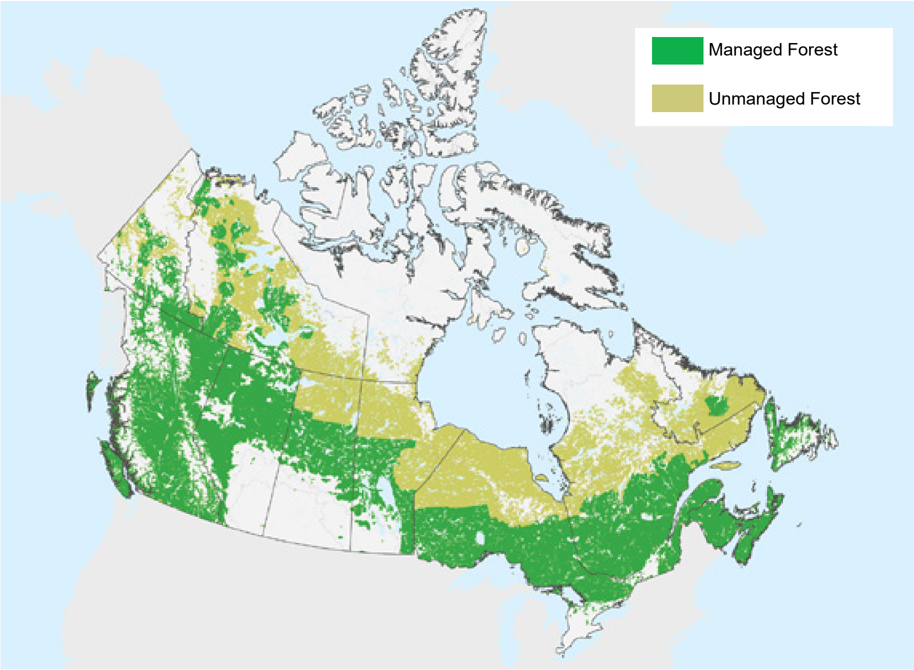
any other land in the Forest Land category where there is active management for timber or non-timber resources, as well as forest areas where there is intensive protection against natural disturbances. All these layers are aggregated and intersected with underlying forest inventory data. The procedures are documented in Stinson et al. (2011).

The model tracks managed forest lands disturbed by harvesting before and after 1990, lands affected by various natural disturbances since 1990 and lands not affected by any disturbances since 1990. Lands not affected by disturbances since 1990 are broken down into stands originating after harvesting or following stand-replacing wildfires prior to 1990. All areas of land in 1990 that were not identified as being of harvest origin were assumed to be of wildfire origin (given that insect disturbances are not stand replacing). These distinctions are used to separate stands dominated by anthropogenic and natural emissions and removals (see Kurz et al., 2018 and section A3.5.2.6).

Forest management activities are documented in the National Forestry Database[3] and additional information on specific activities is obtained directly from provincial and territorial forest management agencies. The Canadian provincial and territorial governments, whose jurisdiction includes natural resource management, provide essential information—notably detailed forest inventory data, details on forest management activities and practices, disturbance information including prevention or control, regional yield tables (volume/age curves), site indices—and regional expertise (Table A3.5–3). The forest inventory data in Canada's NFI (Power and Gillis, 2006) were used for New Brunswick, Manitoba, Saskatchewan, Yukon and the Northwest Territories. More recent and higher-resolution inventory data were provided by Prince Edward Island, Newfoundland and Labrador, Nova Scotia, Quebec, Ontario, British Columbia and Alberta. A series of "method papers" describe the compilation process for each provincial and territorial forest inventory. Since forest inventory data were not collected in the same years, additional steps were necessary to synchronize the inventory data to the year 1990 (Stinson et al., 2011).

Activity data for the burning of harvest residues are obtained from the National Forestry Database for all regions except specific areas of British Columbia where expert opinion is used.[4] In British Columbia, these data refer to slash burning (broadcast burning, slash pile burning or a combination of both), but in other regions of Canada, may also include prescribed burning for fuel control.

Figure A3.5–3 **Managed and Unmanaged Forests in Canada**



3 National Forestry Database, available online at http://nfdp.ccfm.org/en/index.php

4 In British Columbia, slash burning is used on 15% of clear-cut areas on the coast and on 50% of clear-cut areas in the rest of the province, according to expert opinion.

Harvesting firewood for residential heating is a common practice in Canada, with an estimated 93% of annual firewood volumes taken from forested lands and the rest from agricultural woodlands and urban trees. To estimate the impact of this activity on the C balance of Canadian forests, a set of rules was developed to disaggregate these volumes into several components (Hafer et al., 2020; Doyon et al., 2019): softwood, hardwood and mixedwood harvested from forested lands; woody biomass harvested from croplands; urban trees from settlement lands; pellets; and manufactured logs (the last four components were modelled under HWP, see A3.5.3). For the targeted forested land components, a list was developed based on survey data for the spatial analysis units within each RU from which the firewood is assumed to be harvested.

Regional firewood harvesting practices in Canada were represented in CBM-CFS3 and regionally-differentiated disturbance matrix parameters were input in the model for three conceptual firewood harvesting zones: (i) mixedwood-Acadian, comprising the Atlantic Maritime and Mixedwood Plains ecozones; (ii) agricultural, comprising the Sub-humid and Semi-

Table A3.5–3 **Main Sources of Information and Data on Managed Forests**

Description	Source	Spatial Resolution	Temporal Coverage	Reference
Climate data	CFS	Analysis units	1961–1990 normals	McKenney et al. (2001)
Forest inventories and merchantable volume data[a]	Canada's National Forest Inventory (CanFI)	CanFI grid cell	1949–2004	Power and Gillis (2006)
	NL	Analysis units	1991–2006	Provincial experts
	PE	Analysis units	2000	Provincial experts
	NS	Analysis units	2006	Provincial experts
	QC	Analysis units	2000	Provincial experts
	ON	Analysis units	2000	Provincial experts
	AB[b]	Analysis units	1949–1999	Provincial experts
	BC	Analysis units	2011	Provincial experts
Conventional harvest data[c]	National Forestry Database	Provincial boundaries	1990–2021	http://nfdp.ccfm.org/
Prescribed burning	National Forestry Database, for jurisdictions other than BC	Provincial boundaries	1990–2021	http://nfdp.ccfm.org/
Slash burning	BC	Sub-provincial boundaries	1990–2021	Provincial experts
Residential firewood harvest data	Energy sector data for residential firewood use	Reconciliation uUnits	1990–2021	Sections A3.1.4.1 and A3.5.3
Industrial fuelwood data	Energy sector data for industrial fuelwood use	Provincial boundaries	1990–2020	Sections A3.1.4.1 and A3.5.3
Wood waste incineration data	Waste sector data for wood waste	Provincial boundaries	1941–2021	Sections A3.6.3 and A3.5.3
Insect data	Forest Insect and Disease Survey	Spatially explicit	1990–2017	Atlantic Forestry Centre and Pacific Forestry Centre
	Insect data	Spatially explicit	2000–2003	Provincial experts
	QC	Spatially explicit	1990–2021	Provincial experts; https://www.donneesquebec.ca/recherche/fr/dataset/donnees-sur-les-perturbations-naturelles-insecte-tordeuse-des-bourgeons-de-lepinette
	MB	Spatially explicit	1990–2020	Provincial experts and provincial forest health aerial overview surveys; National Forest Pest Strategy Information System
	SK	Spatially explicit	1985–2013	Provincial experts; National Forest Pest Strategy Information System
	AB	Spatially explicit	1990–2021	Provincial experts; Alberta Forest Health Aerial Overview
	BC	Spatially explicit	1990–2021	Provincial experts; BC Forest Insect and Disease Survey; BC Aerial Overview Survey
	YT	Spatially explicit	1994–2018	Provincial experts; Yukon Forest Health Aerial Overview
	NT	Spatially explicit	1990–2021	Provincial experts; Northwest Territories Forest Health Survey
Fire data	National Burned Area Composite	Spatially explicit	1990–2021	https://cwfis.cfs.nrcan.gc.ca/datamart
Drainage data[d]	QC	Province of Quebec boundaries	1980–1985	Provincial experts; historical records; Hillman (1987); Gillies (2016)
	Ministère des Forêts, de la Faune et des Parcs du Québec	Province of Quebec boundaries	1986–1994	https://mffp.gouv.qc.ca/les-forets/connaissances/statistiques-forestieres
	Ministère des Forêts, de la Faune et des Parcs du Québec	Administrative regions of Quebec	1994–2008	https://mffp.gouv.qc.ca/les-forets/connaissances/statistiques-forestieres
	Ministère des Forêts, de la Faune et des Parcs du Québec	Province of Quebec boundaries	2008–2018	https://mffp.gouv.qc.ca/les-forets/connaissances/statistiques-forestieres

Notes:

a. Forest inventory and merchantable wood volume yield data were obtained from Canada's National Forest Inventory and/or from provincial experts where specified.

b. Alberta's forest inventory database comprises provincial forest inventory data for the province's Forest Management Areas, and CanFI inventory data for the remainder of the managed forest landbase.

c. Given the absence of complete harvest data for the most recent reporting year for all provinces and territories, 2021 harvest data were estimated by assuming them to be equal to 2020 values.

d. No new drainage activity data are available in the Province of Quebec since 2016.

arid Prairie ecozones; and (iii) boreal-montane, comprising all other forest ecozones. Firewood harvesting in both the mixedwood-Acadian and agricultural zones is assumed to occur via light thinning (30% removal), while firewood harvesting in the boreal-montane zone is assumed to occur via clear-cut harvesting (85% removal). The selected forest inventory records for firewood harvesting are disturbed in decreasing order of total snag content, to ensure that a reasonable (though unspecified) proportion of the firewood is harvested as deadwood.

Data on the biomass used as residential firewood are obtained from surveys on residential wood use and origin. Sections A.3.1.4.1.4 and A3.5.3 of this report provide additional information on these surveys and the methodology used to convert consumption and use data to firewood volumes. The forest land areas specifically attributed to firewood harvest are defined by the model based on those volume estimates.

Areas disturbed by wildfires were extracted from the Canadian Wildland Fire Information System's National Burn Area Composite (NBAC) database for the years from 1990 to the current inventory year (Table A3.5–3). The NBAC is a composite of low- and medium-resolution remote sensing data and fire mapping data prepared by the Canadian Forest Service and combined with data from resource management agencies from across Canada. The NBAC provides a complete mapping of wildfires using medium-resolution remote sensing data when available, with data from resource management agencies given second priority and low-resolution remote sensing data used only in the absence of other fire mapping data.

Insect disturbances are monitored by aerial surveys (Table A3.5–3), which record the area impacted by the disturbance and assign an impact severity class that indicates the degree of tree mortality or defoliation. The area of impact is assigned to the appropriate analysis unit and host species within it, and the severity of the impact is reflected in the parameters of the disturbance matrix applied (Kurz et al., 2009).

Areas drained for forestry (Table A3.5–3) in private and publicly owned forests in Quebec are estimated using historical documents, consultations and provincial statistics. Spatial allocation by RU was performed using Quebec statistics.

A3.5.2.6. Quantifying Anthropogenic Emissions and Removals

Interannual variations and trends in emissions and removals from managed forests in Canada are dominated by the impact of wildfires and periodic forest insect outbreaks, making it difficult to detect trends resulting from human actions in the forest (Kurz et al., 2008a, 2018; Stinson et al., 2011; Kurz et al., 2013).

The IPCC does not currently provide default methods for separating anthropogenic emissions and removals from those occurring due to natural disturbances, although it has recognized the issues experienced by some countries in reporting emissions from natural disturbances (IPCC, 2010). Furthermore, the IPCC (2010) has encouraged countries that use Tier 3 methodologies to work towards the development of new approaches that can improve the identification of anthropogenic emissions and removals. CBM-CFS3 now has the capability to track and separate emissions and removals in managed forest stands dominated by the impact of significant and relatively recent natural disturbances that have masked the legacy of human management and affected the commercial value and ecological function of the stand from the rest of the area that is subject to active human management activities.

Forest fires are tracked under the natural disturbance component because human-caused ignitions are responsible for a small proportion of the area burned in Canada (~10%; Hanes et al., 2019) and the role of humans is uncertain in explaining increases or decreases in areas burned. Forest fires have been an integral part of the Canadian landscape for millennia. While fire regimes are driven by complex interactions between climate (through temperature, moisture and wind), fuel and humans (as agents of ignition and fire suppression), recent research has shown that climatic drivers mostly control burned areas in northern temperate and boreal forests of eastern Canada, with extreme fire events occurring regardless of human-caused ignitions (Danneyrolles et al., 2021). The dominance of climatic drivers has also been demonstrated in the temperate and boreal forests of western North America (Holden et al., 2018; Gaboriau et al., 2020), and fuel moisture has been suggested as the most critical driver of burned areas and extreme fire events at the global scale (Bowman et al., 2017; Kelley et al., 2019). About 3% of fires in Canada are responsible for 97% of the area burned (Stocks et al., 2002) and generally occur on just a few critical days with extreme weather conditions that are conducive to fire (Wang et al., 2017). These fires are mainly caused by lightning and occur chiefly in remote areas where fire detection and suppression systems are often delayed, compared with human-caused fires that generally take place in more populated, full-suppression zones.

Because direct human actions can be negative and positive—both as agents of ignition (accidentally or intentionally in the case of management activities) and by actively suppressing fires and controlling fuel—their effects on the area burned are difficult to quantify and separate from other drivers. The current default assumption that human activity has neither a positive nor a negative impact on the natural fire regime across the country avoids introducing a bias that is not supported by long-term data and scientific understanding. Although this methodology separately tracks land areas that are dominated by either natural or anthropogenic influences, the evolving balance between those areas over time will capture long-term trends in the natural fire regime in managed forest areas.

This reporting approach ensures that emissions from stands affected by uncontrollable natural disturbances and the subsequent removals due to the regrowth of these stands are tracked separately from those from commercially mature managed stands, allowing for the improved differentiation of emissions and removals associated with direct forest management actions from non-anthropogenic emissions and removals occurring due to natural disturbances.

The history of management activities and natural disturbances in each individual stand (inventory record) in managed forest areas is used to assign stands to two groups. Emissions and removals are identified as being anthropogenic when (i) a stand's growth trajectory has been significantly modified by human intervention—this definition includes commercial clear-cut and partial harvests, commercial and pre-commercial thinning, salvage logging, site preparation, and rehabilitation of and planting on stands that have undergone stand replacement or partial natural disturbances; and when (ii) regardless of its origin, a stand has attained commercial maturity and therefore is actively considered in forest management planning scenarios (eligible to be scheduled for harvest). Once a stand originating from natural disturbance has reached this age, emissions and removals are switched to the reported category.

In contrast, emissions and removals are identified as resulting from natural disturbances when they originate from stands (i) that have been affected by a stand-replacing natural disturbance up to the period that they reach commercial maturity; or (ii) that have been affected by a partial disturbance resulting in reduced standing biomass until that stand has attained biomass equivalent to pre-disturbance values. Only disturbances causing more than 20% mortality are included in the natural disturbance category.

In the initial implementation of this approach in the 2017 National Inventory Report (ECCC, 2017), a fixed value of 60 years was assumed to generally represent a minimum return period to commercial maturity across Canada. Since the 2018 report (ECCC, 2018), regionally specific return periods based on regional differences in forest management practices, species distributions and stand dynamics have been used.

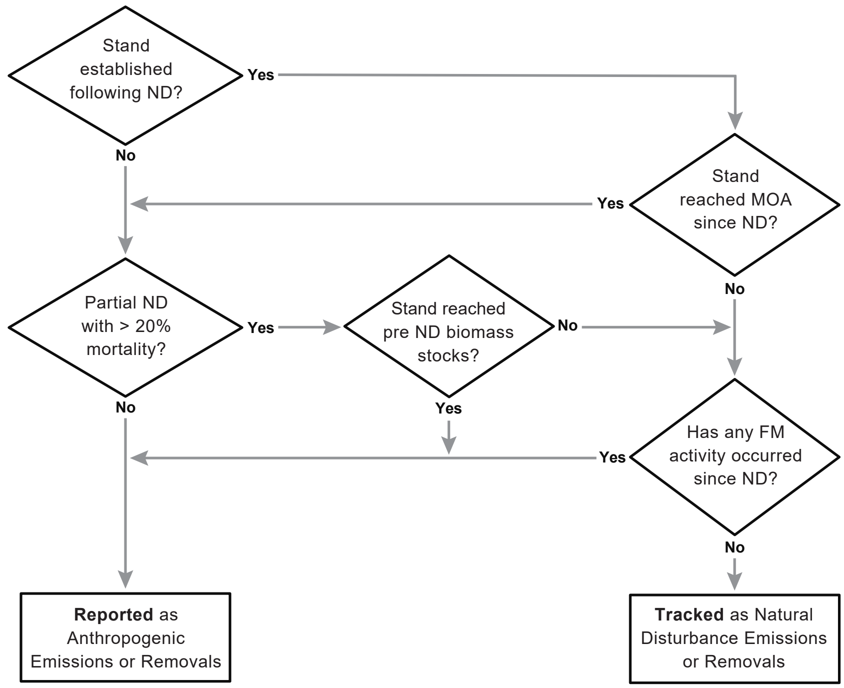
To develop regionally representative definitions of commercial maturity, a questionnaire was distributed to provinces and territories in March 2017. The objective of this consultation process was to document forest management practices across Canada, with a focus on the treatment of naturally disturbed forest stands in operational planning. As such, work with provincial experts provided a minimum return period to commercial maturity ranging from 45 to 99 years, with an average of 76 years. In most cases, provincial agencies defined species-specific commercial maturity based on the maximum mean annual increment of species-specific yield curves for a high productivity site class in a given region. Other provincial agencies used empirical data based on observed regional minimum harvest ages or an age to achieve a specific piece size. On the basis of the species-specific commercial age, a weighted minimum return period was determined for each reporting zone using the proportional breakdown of the commercial species that were attributed a minimum operable age, or minimum harvest age, in that area. Greater detail on the methodological approach used to track anthropogenic emissions and removals can be found in Kurz et al., (2018).

In the current modelling framework, partial natural disturbances occur mainly due to insect infestations. In these cases, above-ground biomass recovery was used to define the recovery period, since the growth trajectory of the stand is only temporarily modified. Stands that undergo insect disturbances causing biomass mortality of 20% or less are not deemed to be dominated by natural disturbances; at this low mortality level, disturbances are considered agents that contribute to stand density reductions.

Stands in which emissions and removals are dominated by natural disturbance dynamics are tracked by querying model results based on a decision-tree approach in which key decision points are based on stand origin, type of disturbance (partial or stand replacing) and an annual assessment of post-disturbance status, based on whether the commercial maturity threshold or pre-disturbance biomass has been reached (Figure A3.5–4).

After exclusion of the non-anthropogenic emissions and removals, the final reported values represent all forest stands in the managed forest land base that have attained commercial maturity or have had their growth trajectory modified by a direct anthropogenic management action in the forest. The area temporarily excluded from reporting in any given year remains relatively constant, within a variation of +2.9/-3.6 million hectares (Mha), as stands undergoing natural disturbance in a given year are removed from reporting and lands that were disturbed historically re-enter reporting. The sum total of each of the stand categories included and excluded is equivalent to the sum of emissions and removals quantified using the methodological approach for reporting total emissions from the managed forest in inventory submissions, prior to implementing the partitioned reporting approach.

Figure A3.5–4 **Decision Tree for Differentiating Between Emissions and Removals of Anthropogenic and Natural Origin**



Notes:
ND = Natural disturbance
MOA = Minimum operable age
FM = Forest management

A3.5.2.7. Land Converted to Forest Land

Records of land conversion to forest land in Canada were available for 1990–2002 from the Feasibility Assessment of Afforestation for Carbon Sequestration initiative (White and Kurz, 2005). Conversion activities for 1970–1989 and 2003–2008 were estimated based on activity rates observed in the data from this initiative. Additional information from the Forest 2020 Plantation Demonstration Assessment was included for 2004 and 2005, and an environmental scan was performed to identify additional sources of information on afforestation rates from 2000 to 2008. Additional afforestation activity data representing 11,194 ha of afforestation events from 2007-2017 were obtained through a data-sharing agreement with Forests Ontario (Magnus et al., 2021).

For jurisdictions other than Ontario, no afforestation activity data for the period 2008-2020 have been identified. Changes in land-use dynamics in Canada during this period suggest that it is not reasonable to assume that afforestation activities would be similar to those before 2008. For the year 2021, an additional 1,435 ha of afforestation events were included as an initial estimate of recent afforestation activities in Canada that was prepared based on a methodology developed for incorporating pre-planting C stocks into the estimation of the initial C content of the events (Hafer et al., 2022).

Each event, regardless of date, source, type or location, was converted to an inventory record for C modelling purposes. All events were compiled in a single data set on afforestation activities in Canada from 1970 to 2021. For 1990–2021, the area planted was stratified by ecozone, province and tree species. The total area planted by province and ecozone, in conjunction with the proportion of species planted in each province, was used to calculate the area planted by species, resulting in estimates of the area converted to forest, by species, in each RU.

Yield curves are not always available for some plantation species or growing conditions (stocking level or site history); those used to estimate growth increments were taken from a variety of sources, including directly from provincial experts and a growth curve compendium prepared for use in Drever et al. (2021). Growth curves for the Forests Ontario plantation sites were developed using the Forest Vegetation Simulator Ontario (Woods and Robinson, 2007), which is a variant of the United States Forest Service's simulator that is adapted for use in Ontario. For all afforestation plantings, species without a yield curve were assigned the yield curve of another species with similar growth characteristics or the species most likely to have been historically present in that area. It was assumed that no woody biomass was present on the site prior to afforestation. Changes in soil C stocks are highly uncertain. The limited time frame of this analysis and the scale of the activity relative to other land use and land-use change activities suggest that the impact of this uncertainty is minimal.

A3.5.2.8. Estimation of Carbon Stock Changes, Emissions and Removals

At the beginning of each annual time step and when an afforestation or forest conversion event is processed, the CBM-CFS3 first assigns the new land-use classification before the impacts of the event are recorded to ensure that the impacts of land-use change (conversion to forests and conversion of forests) are reported in the new land-use category. The selection of forest stands affected by disturbances, whether related or not to land-use change, is based on eligibility rules (Kurz et al., 2009).

Once the model has computed the immediate effect of disturbances on all forest stands, it simulates forest growth, litter fall, and biomass turnover and decomposition, as well as the associated C transfers (annual processes) for all records (managed forest, land converted to forest and land converted from forest), including both stocked and non-stocked stands. The model output consists of C stock changes, fluxes and immediate emissions from burning, allowing the net GHG balance of managed forests to be calculated. Components of fluxes include growth, immediate emissions due to disturbances (C stock changes, and C losses to the atmosphere and to forest products), and decay of both DOM and soil organic matter, including on stands affected by disturbances. During this stage, inventory records that have been in the categories involving land converted to other land uses for 20 years (10 years for land converted to reservoirs) are transferred to the category for land remaining in its final land classification, and the simulation of C dynamics—usually decay—continues in this new category.

The same data outputs are available for converted forest lands (except tree growth), but are reported in the new land subcategories—e.g. the Forest Land Converted to Cropland (CRF Table 4.B subcategory 2.1), Land Converted to Wetlands (CRF Table 4.D subcategories 2.1 and 2.2.1) and Forest Land Converted to Settlements (CRF Table 4.E subcategory 2.1) subcategories. Exceptions consist of estimates of soil organic matter emissions from the conversion of forest land to cropland and from peat extraction sites, which are performed separately; methods are described in sections A3.5.4.3 and A3.5.6.1. Similarly, methods for estimating emissions (as opposed to C stock changes) from the conversion of forest land to flooded land are described in section A3.5.6.2, while methods for estimating emissions from the use and disposal of forest products are described in section A3.5.3.

A3.5.2.9. Uncertainties

Good practice recommends the use of numerical methods for assessing uncertainties within complex modelling frameworks with multiple interactions between data and parameters. These methods are data intensive and computational requirements can quickly become a limiting factor. Not all model parameters or input data have equal influence on model outputs. Careful consideration must therefore be given to balance available computing capacity and the inclusion in the uncertainty assessment of input data, parameters and other functions with a significant influence on model outputs.

There are two approaches to uncertainty assessment that are used: full calculation and extrapolation. Full calculation is a computationally intensive process that involves Monte Carlo simulations, as described in Metsaranta et al. (2017). This process was conducted for the 2018 and 2021 submissions. In the intervening submission years (including the current submission 2023), statistical extrapolation approaches are used instead. This involves a series of multiple linear regression models developed from a full Monte Carlo analysis, using the reported value for a category and the reporting calendar year to predict the uncertainty quantiles of relevance (2.5th and 97.5th). The reporting category values and years from the current submission are then used to estimate these quantiles. The differences between the full analysis quantiles and extrapolated quantiles are typically very small. The decision on whether or not to conduct a full analysis in any given submission depends on a number of factors, including resource availability, and the type and significance of data changes implemented. For the latest six GHG inventory submissions, it has been typical that a full calculation is followed by two extrapolated estimates.

The general approach to uncertainty assessment emphasizes model inputs and parameters as the main sources of uncertainty. The specific uncertainty sources are forest inventory data, influential model parameters and the initialization of soil and DOM C stocks prior to model runs. Additional randomization steps are also included in the development of confidence intervals, by randomly selecting 10 000 bootstrap samples of the output from 100 national-scale Monte Carlo runs (Metsaranta et al., 2017). Not all sources of uncertainty have been captured. Above all, the analysis did not consider the impact of processes that are currently not simulated (Kurz et al., 2013); hence, the results should not be used to assess potential bias (or accuracy) of estimates. The following paragraphs provide details on the characterization of uncertainty sources.

The forest inventory data used in model simulations are compiled for planning and operational purposes. Methods, standards, definitions and quality differ by jurisdiction, depending on its objectives. Although documentation is usually available on the different inventory techniques and procedures used across the country, it seldom contains any quantitative assessment of uncertainty. While it is currently impossible to quantify uncertainties about, for example, managed forest areas, the influence of this uncertainty source can be indirectly built into the uncertainty associated with the biomass increment simulated by the model. For the purpose of this assessment, a 50% uncertainty level is associated with biomass increments. It incorporates uncertainties associated with not only managed forest areas, but also the age-class distribution, yield curves and allometric equations that are used in estimation.

The areas of managed forests affected annually by both natural and anthropogenic disturbances greatly influence forest C dynamics as a whole. Disturbances affect emissions and removals of C in the short term as well as in the long term, through residual decay and age-class distribution. Uncertainties of 10% and 25% are assumed for the areas of managed forests subject annually to wildfires and insect infestations, respectively. The limited total area of forested peatland that was drained suggests that the impact of the uncertainty associated with this activity is minimal.

The uncertainties surrounding the C removed in harvested material are regionally specific and incorporate error ranges for harvested volume (±1%) and standard deviations for the specific gravity of roundwood and the bark adjustment factor (Table A3.5–4). No error was assumed for the C proportion of biomass. The annual coefficient of variation was multiplied by 2 to use a triangular distribution to approximate a normal distribution.

The assessment also provides estimates of the uncertainties associated with emissions due to forest conversion that are subsequently used in the Tier 1 uncertainty reporting for the national estimates in conjunction with the 30% uncertainty for areas converted annually, which is also used in this analysis. The section of this annex on forest conversion describes the derivation of this value (see A3.5.2.10).

Soil and DOM pools contain a considerable amount of C. Previous work has shown that, at the beginning of a complete run, the initial DOM C stocks are sensitive to historical disturbance rates. In this assessment, initial C stocks in the soil and DOM pools were allowed to vary by modifying the historical (pre-1990) fire return intervals. Even though the rates of soil organic matter decay modelled by the annual processes are very low, they do, by virtue of the pool size and forest areas, strongly influence emissions from annual processes.

For the purpose of this analysis, 28 model parameters were allowed to vary in the Monte Carlo runs:

- base decay rates for DOM pools (11 parameters)
- proportion of decayed material that is oxidized, versus that which is transferred to another DOM pool (5 parameters)
- turnover rates for biomass pools (12 parameters)

In the absence of evidence to support more complex functions, all input probability distribution functions for biomass increments, activity data on human and natural disturbances and decay parameters are assumed to be triangular distributions. A gamma probability distribution function is used for fire intervals (Metsaranta et al., 2014).

Significant uncertainty in the modelling framework may result from the random selection of forest stands subject to fire and deforestation disturbances (Kurz et al., 2008b), which interacts with the uncertainty associated with forest inventory data. The random effect of stand selection algorithms is included in the analysis by allowing different seed values to initiate the random selection algorithms.

It is important to note the interactions between input data and parameters. For example, the uncertainty associated with the age of a forest stand (or age-class structure of a forest landscape) may affect the simulation of stand (or landscape) productivity, depending on the yield curves and the particular locations of a given age category along those curves. Emissions due to disturbances—including the conversion of forests to other land categories—are driven not only by the areas affected, but also by the pre-conversion standing C stocks, the parameters of the disturbance matrices that reallocate C among pools or "release" it to the atmosphere, and the post-conversion decay rates. Hence, uncertainties about estimates cannot be obtained from a simple combination of activity data and emission factor uncertainties.

Table A3.5–4 **Uncertainty Ranges for Harvested Carbon, by Canadian Province and Territory**

Province or Territory	Minimum Multiplier	Maximum Multiplier
NL	0.96	1.04
PE	0.88	1.12
NS	0.88	1.12
NB	0.92	1.08
QC	0.86	1.14
ON	0.92	1.08
MB	0.86	1.14
SK	0.92	1.08
AB	0.90	1.10
BC	0.92	1.08
YT	0.84	1.16
NT	0.74	1.26

Note:
Source: Metsaranta et al. (2014)

Uncertainty estimates are developed for reported emissions and removals representing anthropogenic drivers, as well as non-reported emissions and removals due to natural disturbances. In years when no substantial changes occur, a comprehensive uncertainty analysis using Monte Carlo simulation is not performed. Instead, confidence intervals for each category in the current submission are statistically extrapolated for both forest and HWP estimates. These extrapolations use the results of the previous submission, in which numerical estimates of uncertainty were derived using Monte Carlo simulations as explained above and as further described in Metsaranta et al. (2017, 2020). Total uncertainty estimates are allocated to the reported and non-reported categories by utilizing the same categorization procedures used to estimate reported and excluded values (see section A3.5.2.6).

Additional considerations may be warranted to identify the direct human-induced effects, and their uncertainties, on forest C dynamics. Improvements are expected to occur in the coming years, due to increased knowledge, refined procedures, improved computer software implementations and access to more computing capacity.

A3.5.2.10. Forest Conversion

In order to account for the long-term residual effects of forest conversion, annual rates of forest areas converted to other land uses were estimated starting in 1970. The approach to estimating forest areas converted to other land uses before 2010 was based on three main information sources: systematic or representative sampling of remote sensing imagery, records, and expert judgment/opinion. The basic methods were tested in several pilot projects (Leckie, 2006a), and the methodology was implemented across the country. For more recent years, the estimation approach has been based on the representative sampling and mapping of large forest conversion events from remotely sensed imagery only.

The core method involves remote sensing mapping of forest conversion based on samples from Landsat images dated circa 1975, 1990, 2000, 2008, 2013 and 2018. Change enhancements between two dates of imagery are produced to highlight areas of forest cover change and identify possible forest conversion events (i.e., “candidate events”). The imagery is then interpreted to determine whether the land cover of the candidate event was initially forest (at Time 1) and the actual land-use change at Time 2 (Leckie et al., 2002, 2010b). This forest conversion interpretation process is strongly supported by additional spatial data, including: digitized aerial photographs; snow-covered, leaf-off, winter Landsat imagery; secondary Landsat images from other dates and years; ancillary data, such as maps of road networks, settlements, wetlands, woodland coverage, and mine and gravel pit locations; and specialized databases giving locations of oil and gas pipelines and well pads (Leckie et al., 2006; Dyk et al., 2015). When readily available, detailed forest inventory information is also used.

Change imagery is interpreted and analyzed; each forest conversion event larger than 1 ha is manually delineated. The forest type, maturity and density (combined are referred to as the pre-type) prior to forest conversion is interpreted,[5] and the post-deforestation land use recorded (“post-class”). Confidence ratings on the land use at the initial time and a later time period are used in subsequent quality control and field validation procedures.

Monitoring of forest conversion activity covers all forest areas of Canada and is not limited to the managed forest. The entire forested area of Canada is broadly stratified into regions of expected forest conversion level and dominant cause, which dictate the target sampling intensity. Depending on the expected spatial patterns and rates of forest conversion, sampling approaches range from complete mapping to systematic sampling over the entire analysis unit of interest to a representative selection of sample cells within a systematic grid. For example, in populated areas of southern Quebec, in the Prairie fringe and in British Columbia a 12% sampling rate in earlier time periods was generally achieved, with 3.5 km by 3.5 km sample cells at the nodes of a 10 km by 10 km grid (Figure A3.5–5). A lower sampling rate is used in some of the forest activity zones characterized by low population density, where the main economic activities are forestry and other resource extraction. Special cases of known, localized and large forest conversion activities are also identified, such as hydroelectric reservoirs and oil sands development in Alberta. In such cases, the entire areas are handled as single events (“Hot Spot” in Figure A3.5–6), with spatially complete mapping.

In practice, resource constraints limit the size of the remote sensing sample. Wherever possible, a target sampling rate of 12% or 6% was achieved. It is also important to note that different sampling rates may be applied for each time period in an effort to track differing activity rates between time periods. The total areas, either fully mapped or sampled, cover a large portion of Canada’s approximately 346 Mha land base. This total area was mapped over different time periods, of which over 17 Mha were mapped for 1975–1990, 41 Mha were mapped for 1990–2000, 22 Mha were mapped for 2000–2008, 23 Mha were mapped for 2008–2013 and 15 Mha were mapped for 2013–2018 (Figure A3.5–6). Mapping of sampled areas and larger individual events is updated on a roughly five-year cycle and may be progressively integrated to the monitoring system for the more recent time periods.

Areas of northern Canada were previously unrepresented in the National Deforestation Monitoring System estimate due to the anticipated low activity; consequently, zero deforestation had been assumed in this region (Dyk et al., 2015). In order to verify activity levels in the north, a full-coverage deforestation mapping project in the Northwest Territories’ Taiga Plains

5 See Chapter 6 for the definitional parameters of “forest.”

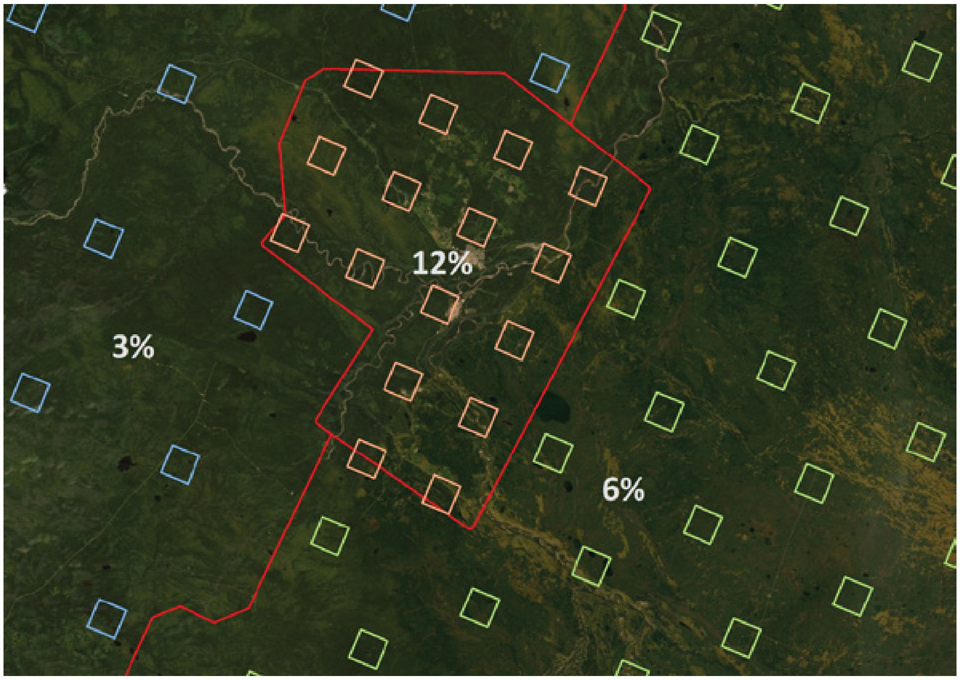
ecozone (RU50) was undertaken based on existing anthropogenic disturbance data sets. RU50 was processed with full area mapping (Figure A3.5–6), which showed that deforestation in this reconciliation unit represents less than 0.2% of Canada's annual deforestation rate. These results started to be reported in the 2022 submission.

Records data were also gathered when available, consisting mostly of information on forest roads, power lines, oil and gas infrastructure, and hydroelectric reservoirs (Leckie et al., 2006). The temporal coverage, availability and applicability of these records are assessed to determine the most appropriate information sources (records or imagery). Records data are sometimes used to aid in the validation of estimates made through image interpretation. In particular, records data were used in the early mapping efforts in British Columbia to provide estimates of conversion activity for power lines and oil and gas operations. The interpretation of remote sensing imagery is used to assess the areas of forest converted as a result of hydroelectric development.

Expert opinion is only called upon when remote sensing based sampling is insufficient and records data are unavailable or of poor quality. Expert judgment is also used to reconcile differences between records and remote sensing information and to resolve major sequential time periods of mapping (i.e. 1975–1990, 1990–2000, 2000–2008; the most recent time periods are measured on a roughly five-year cycle). In such cases, available expert opinions and data sources are compared, remote sensing and records data are reviewed, and decisions are made (Leckie, 2006b; Leckie et al., 2010a; Dyk et al., 2015). In the case of most current estimates—and certainly those with a significant impact—estimates are derived directly from remote sensing samples.

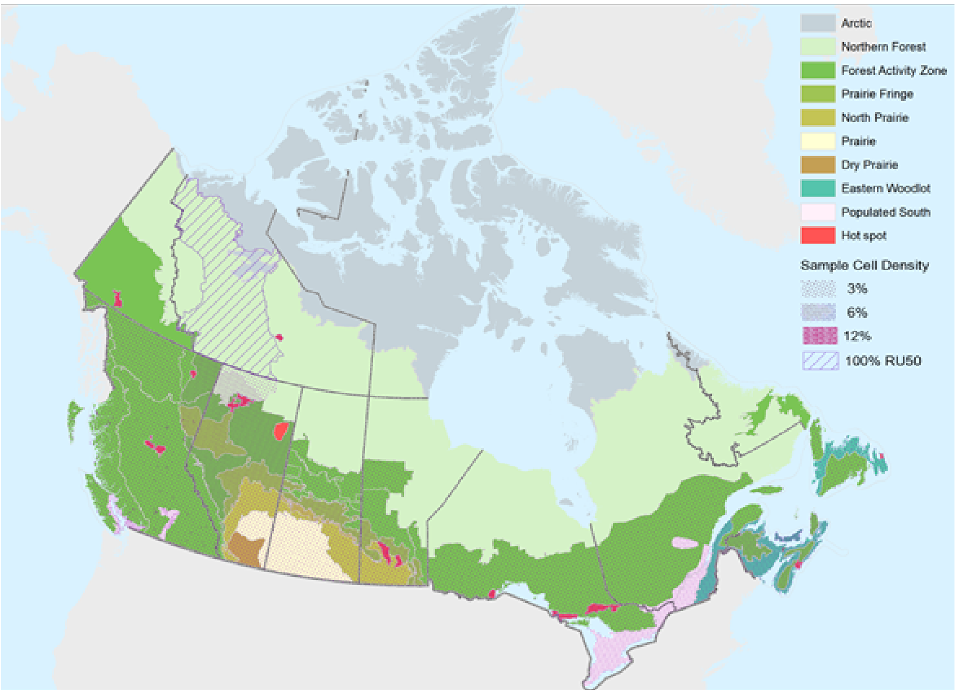
The activity data are compiled and summarized initially by analysis unit and all the conversion events are assembled in a database. The purpose of this compilation is to summarize events to obtain detailed post-conversion classes for each RU. This compilation process also involves the insertion of records data and expert judgment. During these procedures, a local forest conversion rate (ha year^{-1}) is obtained from the compilation of events, based on the time interval between the images. Since the available imagery was not necessarily dated to a specific year, the rates cover different time periods. During the data compilation phase, forest conversion events are assigned to a time period, and the corresponding rate of forest conversion is assigned to that period. For example, a 7.0-ha event encountered on imagery from the period 1975–1989 would yield a 0.5 ha year^{-1} rate (7.0 ha/14 years) and then would be assigned to the period 1975–1990. The total area interpreted in an analysis unit for that time period is then used to determine the relative rate of forest conversion (ha year^{-1} per km^2 interpreted) for all events of the same type. Relative rates are scaled up for each analysis unit. Finally, data are grouped by end land use (e.g., agricultural crop or rural residential) and, in turn, are summarized by broader categories when recompiled by RU.

Figure A3.5–5 **Three Satellite Imagery Sampling Rates for Forest Conversion Mapping**



Note: Background imagery: Area near Fort Nelson, British Columbia (ESRI World Imagery). The denser grid cells at the centre represent a 12% sampling density; the lighter grid on the right, 6% intensity; and the sparse grid on the left, 3% intensity.

Figure A3.5–6 **Forest Conversion Strata and Areas Sampled in 2013–2018**



The remote sensing data were derived from medium-resolution imagery from circa 1975, 1990, 2000, 2008, 2013 and 2018, and more recent years as new imagery has become available, while the records data are annual or summarized over a given time period. As explained, to date, the core remote sensing method has provided five distinct average rates of forest conversion in the mapped time periods, but no annual estimates of these rates. The preparation of annual forest conversion rates from 1970 to the current inventory year would require the simultaneous application of two procedures: (i) the extrapolation of annual rates prior to 1983 and beyond the midpoint of the latest mapped time period available; and (ii) linear interpolation between the midpoints of the mapped time periods and recent analyses completed at the time of submission (Figure A3.5–7). Added to the interpolated data are individual large events for which actual disturbance information is known, either from records information or a detailed mapping activity, for example, hydroelectric reservoirs.

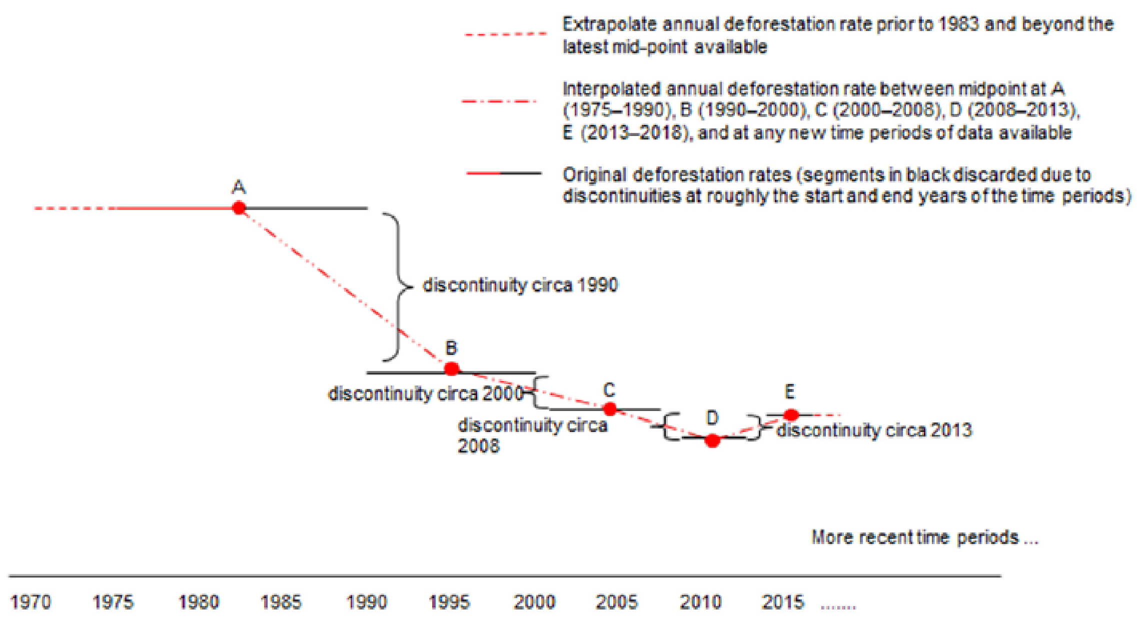
Quality Assurance / Quality Control of Forest Conversion Data

Great care was taken in analyzing the records data, their suitability and their limitations. Their documentation was examined, personnel involved in managing and implementing data collection and storage were interviewed and, when available, numbers were checked against independent data sources, the high-resolution imagery sampled and expert knowledge.

The interpretation of remote sensing imagery follows defined procedures (Leckie et al., 2010b; Dyk et al., 2015), although it is conducted by a variety of organizations, including provincial government forestry or geomatics groups, remote sensing or mapping companies, research and development organizations and in-house government staff. The basic quality control process for image analysis includes internal checks within the mapping agency or company by a senior person; real-time quality assurance by Canadian Forest Service specialists during interpretation, with feedback provided within days of the interpretation of an area; and the final quality assurance and vetting of the interpretation by the Canadian Forest Service. Field validation is conducted on an ongoing basis as resources permit. Each quality control point and revision is documented in the geographic information system (GIS) database of conversion events (Dyk et al., 2015).

Records of decision as to data used and expert judgement applied, as well as decisions on the resolution of contradictory data, are documented within the overall processing database (Leckie, 2006b) and updated for each new submission (Dyk et al., 2015). Data sources and limitations are recorded, and remote sensing data and interpretations archived.

Figure A3.5–7 **Procedure for Developing a Consistent Time Series of Forest Conversion Rates**



Uncertainty of Forest Conversion Data

The development of an uncertainty estimate for forest conversion data is a complex and difficult task because of the spatial and temporal variability of the data. Compared with earlier estimates, the current estimates benefit from several years of experience and knowledge gained from the development of previous estimates (Leckie, 2011; Dyk et al., 2015). Specific improvements include:

- expanded data sets with additional Earth observation (EO) data; Landsat, Sentinel 2, SPOT-5 and high-resolution satellite imagery; and aerial photographs;
- expansion of the sampled area in targeted and other areas;
- analysis and validation of records data using high-resolution imagery (for example, co-disturbances from pipelines and access roads);
- extension of temporal coverage to the most recent time period;
- review of the 1970–2004 deforestation time series based on more current spatial analysis;
- validation of existing anthropogenic disturbance data sets for northern Canada;
- better knowledge resulting from the increased experience and expertise gained from quality control review and validation activities.

These improvements result in enhanced detection, delineation and determination of event size and cause, as well as a more accurate estimate of timing of conversion events.

Two approaches to estimating uncertainties were considered: an empirical and an analytical approach. The resulting estimate is based on the consideration of these approaches and provides an estimate of uncertainty associated with activity area estimates. Additional sources of uncertainty related to the forest type being converted, the post-conversion land category and event timing are not considered.

The empirical approach is an attempt to assess the overall uncertainty in the forest conversion area estimate. This approach provides an estimate of uncertainty that considers all the varied components of uncertainty and their potential interactions.

The empirical estimate of uncertainty was developed by estimating the extreme low, low, high and extreme high forest conversion rates for each RU and end-use class, based on expert knowledge of activities and practices at a regional scale. All these estimates were then compiled on a national basis. Comparisons between extreme and non-extreme estimates provided insight into the possible range of conversion activity. Based on this exercise, the estimate of the overall uncertainty associated with forest conversion was determined to be in the range of ±20% to ±30%.

The analytical approach breaks down uncertainty into subcomponents and then combines them through simple error propagation. The components considered are omission, commission, sampling and boundary delineation errors.

Omission and commission errors are influenced by a number of factors, but in particular by the date and quality of the pre- and post-imagery. Throughout the time series, omitted events tend to be smaller in size, while commission errors usually result from a misinterpretation rather than an oversight, and thus are less size-dependent. Commission and omission errors tend to offset each other. For the post-2000 time periods, commission errors are likely to be greater than omission errors, particularly because of an insufficient time lapse/period after disturbance to confirm that areas are in fact permanently deforested.

The uncertainty associated with boundary delineation errors involves the errors resulting from the displacement of the event boundary from its actual or true boundary. Both underestimation and overestimation of the area can result. This source of uncertainty is greatly influenced by the quality and resolution of the imagery used in the delineation process; improvements in resolution and image quality reduce this source of uncertainty.

Estimates of sampling uncertainty focus on the sampling process and the scaling of estimates to large regions (strata/RU). The sampling process is a mixture of wall-to-wall mapping and systematic sampling. In some areas, the sample coverage and design was different in each mapping period. The sample error depends on the amount of activity in each region within each time period sampled. In addition, it is dependent on the conversion event size and spatial distribution (Leckie et al., 2015). Uncertainty due to sampling and scaling activity is therefore regionally variable and, because the causes of conversion activities may differ by region, the uncertainty is variable overall.

The results of this analytical approach are consistent with those obtained using the empirical approach. The result was a conservative estimate, which sets the uncertainty at the higher range of ±30%. Further work will help improve the current understanding of the various sources of uncertainty, their interaction and the approaches used to combine these components.

The ±30% range is an overall estimate considering all time periods, regions and forest conversion types. Caution should be exercised in applying this range to the cumulative area of forest land converted to another category over the last 20 years, or 10 years in the case of reservoirs (land areas reported in the CRF tables).

A3.5.3. Harvested Wood Products

The LULUCF sector of the inventory includes an estimate of the CO_2 emissions associated with the use and disposal of HWP manufactured from wood resulting from forest harvesting (including firewood) and forest conversion activities in Canada and consumed either in Canada or elsewhere in the world, in accordance with the general framework of the simple decay approach, as described in the Annex to Volume 4, Chapter 12, of the 2006 IPCC Guidelines (IPCC, 2006). This approach is similar to the production approach, but differs from it in that the HWP pool is treated as a C transfer related to wood harvest, and hence the instant oxidation of wood in the year of harvest is not assumed. The approach tracks the fate of C in all woody biomass harvested domestically and taken off-site. Emissions of CO_2 from HWP use and disposal are estimated and reported in the LULUCF sector, while CH_4 and N_2O emissions from HWP combustion or domestic decomposition are estimated and reported in the Energy and Waste sectors.

General Approach and Methods

A country-specific model, the National Forest Carbon Monitoring, Accounting and Reporting System for Harvested Wood Products (NFCMARS-HWP), was developed to estimate and report on the fate of C in the wood harvested in Canada's forests and other wooded lands.

Model Inputs and Data Sources

Inputs to the model include the annual mass of C transferred to forest products that result from conventional forest harvesting in forests, from residential firewood harvesting in forests and other wooded lands and from forest conversion activities since 1990. This input is spatially distributed by RU (see section A3.5.1), as calculated by CBM-CFS3 (see section A3.5.2.1), thus ensuring there are no gains or losses as C flows from wood producing lands to products.

Data on the annual volume of residential firewood and industrial wood waste used for bioenergy are provided by the Energy sector. Residential firewood consumption data were collected in a survey on residential wood use for the years 1997, 2003, 2007, 2015, 2017 and 2019 (Statistics Canada, 1997, 2003, 2007, 2015, 2017, 2019). Pellet and manufactured log consumption data were collected for the years 1996, 2006, 2012, 2017 and 2019 (Canadian Facts, 1997; TNS, 2006, 2012; Statistics Canada, 2017, 2019). These data were obtained for the provinces only (i.e. not the territories) and grouped by major appliance category (i.e. advanced and conventional fireplaces and fireplace inserts, wood stoves, wood furnaces, pellet stoves, hydronic heaters, water heaters and other equipment). The 2017 and 2019 surveys also gathered data on the type of wood used for firewood, which were spatially aggregated by RU (Trégaro, 2020). As a result, species-specific wood densities could be used (Blondel and Tracey, 2018), and were maintained constant throughout the time series. Pellet and manufactured log consumption data were collected on a mass basis. These data were interpolated and extrapolated to other years using the number of heating degree-days in each province in relation to the survey years. Data on firewood consumption in the territories come from fuelwood and firewood harvest statistics from the National Forestry Database,[6] while data on the industrial consumption of fuelwood (biomass and spent pulp liquors) for energy production come from the annual Report on Energy Supply and Demand in Canada (RESD). See section A3.1.4.1.4 for more details.

Data on wood waste incinerated in controlled incineration facilities are provided by the Waste sector, details on specific data sources and methodologies used are provided in section A3.6.3.

Carbon input from historical harvests is derived from national-level commodity production data from Statistics Canada covering the 1941–1989 period. For the 1900–1940 period, C inputs are back cast based on historical production data by extrapolating information from the 1941–1989 period, while consumed and exported quantities are calculated using average proportions from the five-year period from 1961 to 1965.

Model Flow and Parameters

The model uses a conceptual flow network describing the movement and transformation of harvested wood (Figure A3.5–8). The model takes the C inputs and, in annual time steps, exports some of the harvested roundwood, converts all harvested wood to commodities (sawn wood and other industrial roundwood, wood-based panels, pulp and paper, pellets and manufactured logs used for bioenergy, and residuals referred to as "milling residue"), exports some of the commodities produced, and keeps track of the additions to and retirements from HWP in use or combusted for bioenergy. The complete model consists of 15 of these networks—one for each province and territory (except Nunavut), plus one each for the United States and Japan, and one that combines all other countries importing Canadian wood products. The on-site decay of harvest residues continues to be captured in C stock changes in the DOM pool of the Forest Land category.

Recent statistics available in two FAO databases, Forestry Production and Trade[7] and Forestry Trade Flows,[8] were used to determine the proportion of Canadian roundwood and commodity production exported to three main destinations. For example, according to current statistics from the FAO, in any given year, around 98% of domestically harvested industrial roundwood remains in Canada for further transformation, of which about 70% is converted to sawn wood, wood-based panels, other industrial roundwood, or pulp and paper products. Similarly, over the entire time series, roughly 33% of sawn wood, between 19% and 65% of wood-based panels and less than 13% of pulp and paper are used domestically. The proportion of HWP transferred from the in-use pool is determined by applying Equation 12.1 from Volume 4, Chapter 12 of the IPCC 2006 Guidelines (IPCC, 2006). All C retired from the in-use pool is assumed to be instantly oxidized. Emissions from residential firewood use and industrial processes fueled by milling residue (e.g. industrial bioenergy) are represented separately to prevent any potential overlap with estimates reported by the Energy sector.

Manufacturing efficiencies determine the proportion of industrial roundwood biomass converted into commodities, with the unused fraction treated as milling residue. These proportions are calculated using a mass-balance approach that reconciles the domestic harvest with FAO data on commodity production and trade. Manufacturing efficiencies are calculated annually for each commodity type: separately for Canada, the United States and Japan and jointly for all other export destinations. Default bark expansion factors and wood C content were used for all countries (Table A6.5–1). Default parameters were used to convert product volumes to units of C for countries other than Canada and the United States and when country-specific parameters were not available for Canada or the United States (Table A6.5–2). Canada-specific wood density values were used for domestic roundwood, sawn wood, panels and other industrial roundwood, and default values were used for domestic pulp and paper. Country-specific values were used for all domestic quantities for the United States. Default values were used for domestic and imported quantities for Japan and elsewhere. It is assumed that all wood fibre feedstock produced in a given year is processed by the forest products manufacturing sector in the same year.

All wood transferred from the forest to the HWP pool is included in the HWP model, but some of the products associated with portions of the wood, such as wood chips and pellets, are not explicitly identified in the data. Data on chips and pellets are estimated from firewood consumption surveys, unlike the data on other HWP. Wood used for bioenergy, which includes pellets and chips, is assumed to be sourced from the milling residue output category in the HWP model (see Figure A3.5–8). This C is quantified and allocated to bioenergy but is undifferentiated from other residual waste, all of which is assumed to be oxidized on disposal. The export of wood chips/pellets is currently not considered in the model.

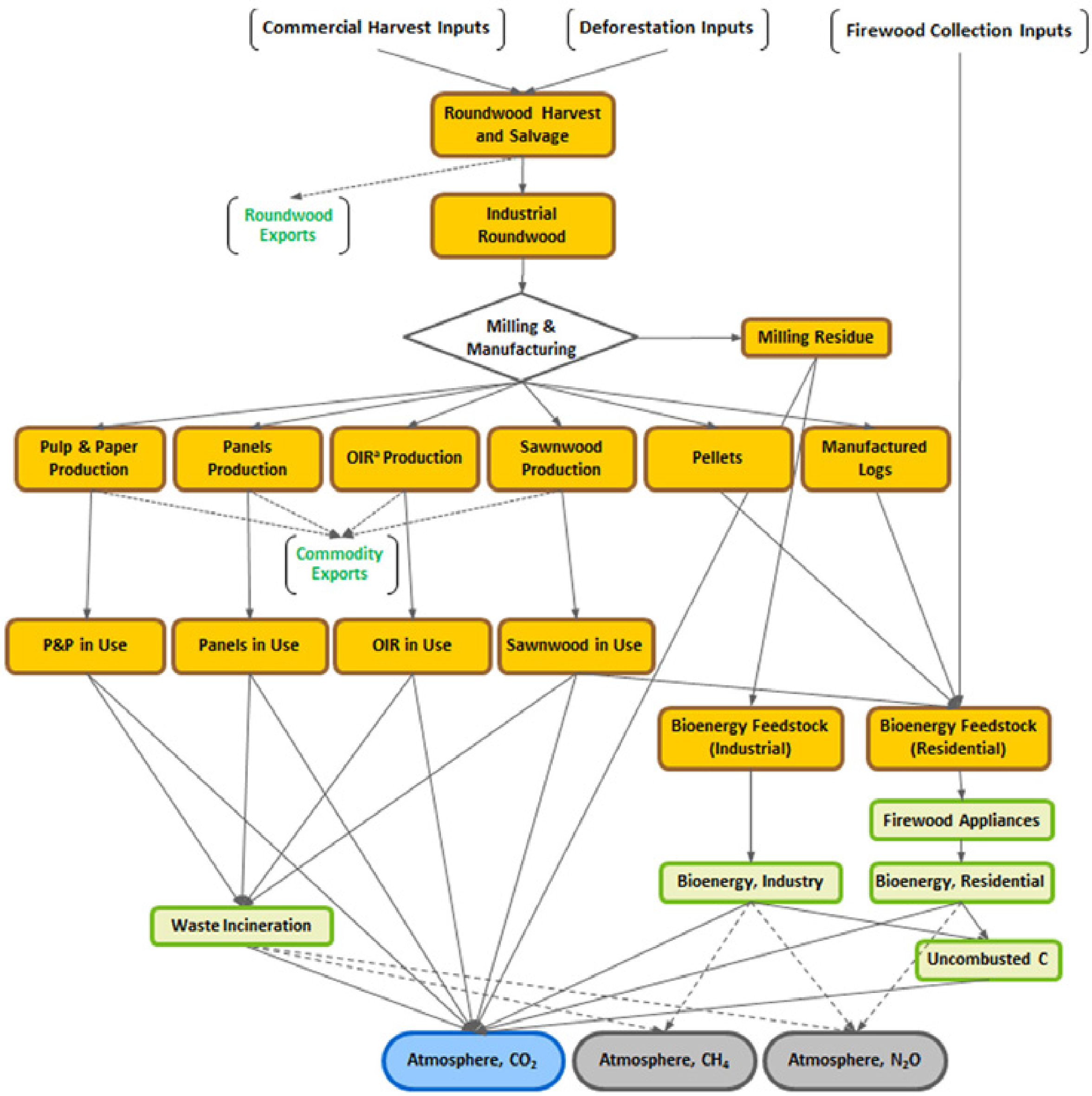
The model starts the pool in 1900 and applies product-in-use half-life parameters to wood product types based on geographic location. Half-life parameters are sourced directly from Table 3a.1.3 of IPCC (2003) or derived from the same table using production-weighted averages corresponding to the wood product categories in the NFCMARS-HWP (Table A6.5–3).

6 National Forestry Database, available online at: http://nfdp.ccfm.org/en/data/harvest.php.

7 FAOSTAT Forestry Production and Trade, available online at: http://www.fao.org/faostat/en/#data/FO.

8 FAOSTAT Forestry Trade Flows, available online at: http://www.fao.org/faostat/en/#data/FT.

Figure A3.5–8 **A Simplified Schematic of Carbon Flows in Harvested Wood Products**



Note:
a. OIR = Other Industrial Roundwood

Biomass Combustion

Biomass emissions reported in the Energy sector are grouped into three main categories based on the source: (i) residential firewood; (ii) industrial wood wastes (including spent pulp liquors); and (iii) fuel ethanol/biodiesel (assumed not to come from wood waste or pulp liquors).

Residential firewood combustion produces CO_2, CH_4, N_2O and some remaining unaccounted-for C likely found in VOCs, unburned hydrocarbons and charcoal that are assumed to be instantly oxidized, in amounts that are dependent on the combustion technology used. Emissions are derived by multiplying the amount of wood burned in each appliance type by the emission factor for that appliance type. The relevant emission factors are given in Table A6.6–1, expressed in grams of gas emitted per kilogram of fuel combusted, which for the purposes of the model have been converted to tonnes of C emitted as gas per tonne C in fuel.

Emissions from the industrial use of wood-based energy (treated as milling residue in the model) are assumed to result from the combustion of wood wastes (i.e. hog fuel) and spent pulping liquors by the pulp and paper manufacturing sector. As with residential bioenergy use, emissions from industrial biomass energy use are derived by multiplying the amount of fuel consumed by the emission factor for that fuel type. The emission factors for both industrial wood waste and spent pulp liquors are also given in Table A6.6–1. Note that these emission factors are expressed in grams of gas emitted per kg of fuel consumed, with the fuel assumed to have a moisture content of 0%.

The processing of residential firewood data ensures consistency with the methods used for the Energy sector and that the impacts of this type of harvest on the forest and other wooded ecosystems are adequately represented in land emission modelling at the finest possible spatial resolution (Trégaro, 2020). All biomass C inputs to the firewood pool are based on the annual volumes provided by the Energy sector. Specifically, annual quantities of residential bioenergy consumption (in tonnes C) are calculated for each RU, in each of seven (7) allocation categories: (i) softwood harvested from forest land; (ii) hardwood harvested from forest land; (iii) mixedwood harvested from forest land; (iv) woody biomass harvested from cropland; (v) firewood harvested from urban trees in settlements; (vi) pellets; and (vii) manufactured logs, representing the targets of firewood harvesting to be implemented in the models (Trégaro, 2020; Trégaro and Blondel, 2019; Hafer et al., 2020). Targets for the first three categories in the list were implemented in the CBM-CFS3 simulations (see A3.5.2.5), while targets for the last four categories were implemented in the HWP model. The impacts of firewood harvesting on the Cropland and Settlements land-use categories were estimated (see sections A3.5.4.1 and A3.5.7.1 for more details).

Uncertainty

Uncertainty estimates associated with this category are mainly based on the uncertainty of the C inputs, namely (i) the estimated C in forest products from forest harvest and forest conversion in CBM-CFS3; (ii) the volume of residential firewood provided by the Energy sector; and (iii) statistics available on pre-1990 commodity production.

The uncertainty analysis (Metsaranta et al., 2020) divides the uncertainty into three categories: (i) uncertainties in assumptions and approaches, e.g. the assumption that HWP disposal follows the exponential decay pattern; (ii) uncertainties in factors or parameters that are not derived from activity data, e.g. half-lives of in-use-commodity pools and landfill pools; and (iii) uncertainties in input and allocation parameters that refer to C mass inputs (e.g. roundwood harvest) and partitioning parameters derived from activity data.

A sensitivity analysis was used to filter out parameters in which variations are unlikely to cause significant changes to the emission results prior to the Monte Carlo analysis. Uncertainty distributions and ranges were based on literature where possible and, where no distributions were available, on expert judgment.

Additional parameters were added to the Monte Carlo analysis, including uncertainty distributions for historical inputs (pre-1990 harvest), contemporary inputs (harvest since 1990) and five allocation parameters related to bioenergy that were added to the HWP model structure. Historical inputs are directly allocated to in-use-commodity pools and are varied using a multiplier assigned a uniform distribution with a range between 0.75 and 1.25. Contemporary inputs are obtained from the outputs of CBM-CFS3, which correspond to a range of C mass values. These outputs are used as inputs for the HWP uncertainty analysis. Three sets of pools with their corresponding events and parameters were also added to the analysis: pellets, manufactured logs and bioenergy (residential and industrial). The sample size (n) for the Monte-Carlo runs was 100.

As already noted in A3.5.2.9, in years with no substantial changes, a comprehensive uncertainty analysis is not performed and, instead, confidence intervals for each category in the current submission year are statistically extrapolated using the results of the previous submission.

A3.5.4. Cropland

The methodologies described in this section apply to C stock changes in mineral soils subject to cropland management and to the conversion of land in the Forest Land and Grassland categories to the Cropland category, CO_2 emissions from the cultivation of histosols, changes in the biomass of woody perennial crops, and N_2O emissions from soil disturbance resulting from conversion to cropland. The estimation methodologies for C stock changes and GHG emissions from the biomass and DOM pools resulting from conversion of forest land to cropland are provided in section A3.5.2.10.

A3.5.4.1. Cropland Remaining Cropland

A detailed description of the methodologies used for estimating C changes associated with tillage practices, perennial/annual crop conversion, and forest land and grassland conversion to cropland can be found in McConkey et al. (2007a).

Change in Carbon Stocks in Mineral Soils

Changing Management Practices

The amount of organic C retained in soil represents the balance between the rates of input from crop residues and losses through soil organic carbon (SOC) decomposition. How the soil is managed determines whether the amount of SOC stored in a soil is increasing or decreasing. The development of the CO_2 estimation methodology is based on the premise that, on long-existing cropland, changes in soil C stocks over time occur following changes in soil management that influence the rates of either C additions to, or C losses from, the soil. If no change in management practices occurs, the C stocks are assumed to be at equilibrium, and hence the change in C stocks is deemed to be zero.

A number of management practices are generally known to increase SOC in cultivated cropland, such as reduction in tillage intensity, intensification of cropping systems, adoption of yield-promoting practices and re-establishment of perennial vegetation (Janzen et al., 1997; Bruce et al., 1999). Adoption of reduced tillage (RT) or no-till (NT) can result in significant accumulation of SOC compared with intensive tillage (IT) (Campbell et al., 1995, 1996a, 1996b; Janzen et al., 1998; McConkey et al., 2003). Many cropping systems can be intensified by increasing the duration of photosynthetic activity through the greater use of perennial forage (Biederbeck et al.,1984; Bremer et al.,1994; Campbell et al.,1998). Switching from conservative to conventional tillage or from intensive to extensive cropping systems will generally reduce C input and increase organic matter decomposition, thereby reducing SOC.

VandenBygaart et al. (2003) compiled published data from long-term studies in Canada to assess the effect of agricultural management practices on SOC. More recent analyses (Fan et al., 2017; Liang et al., 2020a,b; Liang et al., 2021) have prompted new methodological developments. The original compendium of data by VandenBygaart et al. and these new studies, as well as activity data available from the *Census of Agriculture*, provide the basis for identifying key management practices and management changes used to estimate changes in soil C stocks. Emissions and removals of CO_2 from mineral soils are estimated for the following land management changes (LMCs):

1. Change in mixture of crop types
    - a) Increase in perennial crops
    - b) Increase in annual crops
2. Change in tillage practices
    - a) IT to RT
    - b) IT to NT
    - c) RT to IT
    - d) RT to NT
    - e) NT to IT
    - f) NT to RT
3. Change in crop productivity / crop residue C input
4. Manure application

Where nutrients are strongly limiting, proper fertilization can increase SOC and, in such conditions, fertilizer or other nutrient-enhancing practices are generally applied. Irrigation in semi-arid areas can affect SOC, but the impact is unclear and the area of irrigated land has been relatively constant in Canada. Therefore, it is assumed that the selected LMCs represent the most important and consistent influences on SOC in mineral soils.

Area-Based Carbon Stock Change Factor

To estimate C emissions or removals, an SOC stock change factor specific to each combination of SLC polygon and management change is multiplied by the area of change. The factor is the average rate of SOC change per year and per unit of area of LMC.

Equation A3.5–1

$$\Delta C = F \times A$$

ΔC = change in SOC stock for inventory year, Mg C

F = average annual change in SOC subject to LMC, or C factor, Mg C ha^{-1} yr^{-1}

A = LMC area, ha

The areas affected by LMCs, such as changes in tillage and crop type, are obtained from the *Census of Agriculture*. Census data provide information on the net change in area over five-year census periods. In practice, land probably both enters and leaves a land management practice, and combinations of management changes occur. However, because only net change data are available, two assumptions are made: the additivity and reversibility of SOC factors. Reversibility assumes that the factor associated with an LMC from A to B is the opposite of that associated with the LMC from B to A. Additivity assumes that C changes from each individual LMC occurring on the same piece of land are independent and therefore additive. This assumption is supported by the findings of McConkey et al. (2003), who reported that the impact of tillage and crop rotations on SOC is additive.

A relatively large set of Canadian observations is available on long-term changes in SOC resulting from LMCs such as the adoption of NT (VandenBygaart et al., 2003; Campbell et al., 2005; Liang et al., 2020). However, even this extensive data set does not cover the full geographical extent of Canadian agriculture. In addition, difficulties arise in comparing measurements among research sites, determining the duration of an effect, estimating full uncertainty from a range of initial soil conditions and determining the variability of soil C stocks without a LMC.

Because of these limitations, a well-calibrated and validated model of SOC dynamics, the Century model (Parton et al., 1987, 1988), was used to derive the individual SOC factors for changes between NT and IT, RT and IT, RT and NT, and annual and perennial crops. The Century model has been widely used to simulate SOC changes under Canadian conditions (Voroney and Angers, 1995; Liang et al., 1996; Monreal et al., 1997; Campbell et al., 2000, 2005; Pennock and Frick, 2001; Carter et al., 2003; Bolinder, 2004).

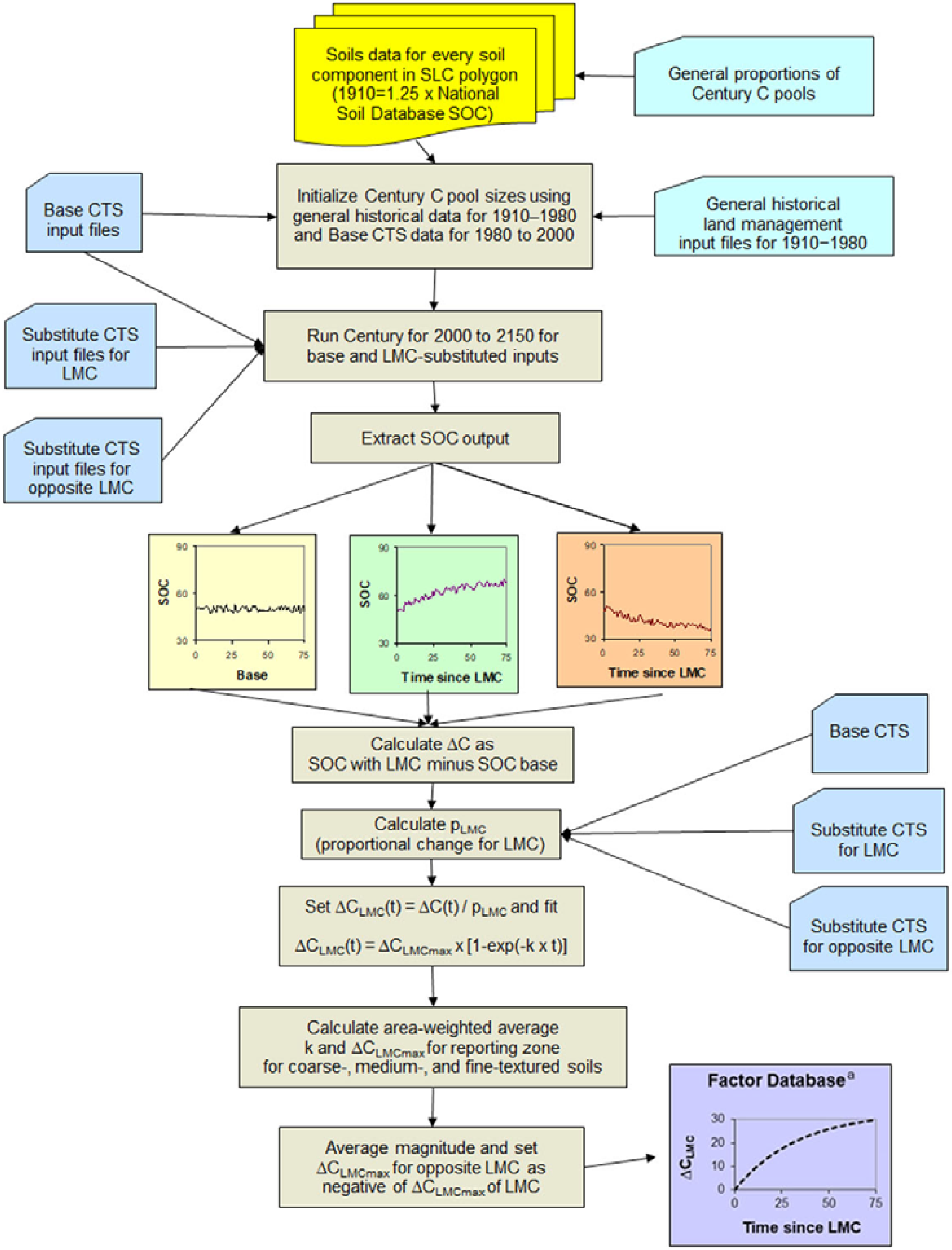
Smith et al. (1997, 2000, 2001) developed an approach using the Century model to estimate SOC changes on agricultural land in Canada. Estimating C changes required the development of a generalized description of land use and management on cropland from 1910 onward for a sample of soil types and climates across Canada. These scenarios were generated from a mixture of expert knowledge and agricultural statistics on land management, including crop types, fallow practices and fertilizer application (Smith et al., 1997, 2000). They were used for the first comprehensive assessment of SOC changes on agricultural land within a broader assessment of soil health (McCrae et al., 2000).

The SOC values in the SLC polygon attribute database (Canadian Soil Information Service, or CanSIS) served as the starting point for developing C factors for tillage and crop mixtures (Figure A3.5–9 and Figure A3.5–10). These SOC values were derived from measurements from soil surveys and land resource studies (Tarnocai, 1997) and are assumed to represent average SOC values on cropland in 1985. The initial SOC in 1910 was considered to be 1.25 times the SOC in the corresponding SLC polygon today Changes in SOC factors were estimated using the difference in SOC stocks over time based on simulation of a generalized land-use and land management scenario with and without the LMC of interest (Smith et al., 2001).

To develop area-based factors, a 10-year crop-and-tillage system (CTS) was developed for each analysis unit and census year, using data from the *Census of Agriculture*. The CTS focused on seven crops or crop types (grain, oilseeds, pulses, alfalfa, root crops and perennial crops) and three tillage practices (IT, RT and NT). Essentially, each CTS represents a mix of crops and tillage practices in space as a mix of crops and tillage practices in time. Under this scheme, a polygon with 20% of cropland area in grain and 20% of cropland area in NT, for example, has 2 of 10 years in grain and 2 of 10 years in NT. Temporal sequences of crop and tillage practices are developed from expert-defined rule-sets, such as “summerfallow never follows summerfallow” and “corn typically follows soybeans.” The construction allows a base CTS and substitutions of LMCs in the CTS to be readily input in the Century model.

The SOC change factor is determined as factor = (C for CTS with LMC – C for base CTS) / [(fraction of CTS substituted with the LMC) × (duration considered)]. If a land management system is defined as a particular mix of crops and tillage practices on a specified land area, a change in SOC due to an LMC (ΔC_{LMC}) can be estimated as the difference in SOC stock between two land management systems divided by the proportion of the land area subject to an LMC.

Figure A3.5–9 **Method for Deriving Area-Based Carbon Factors for a Land Management Change of Interest**

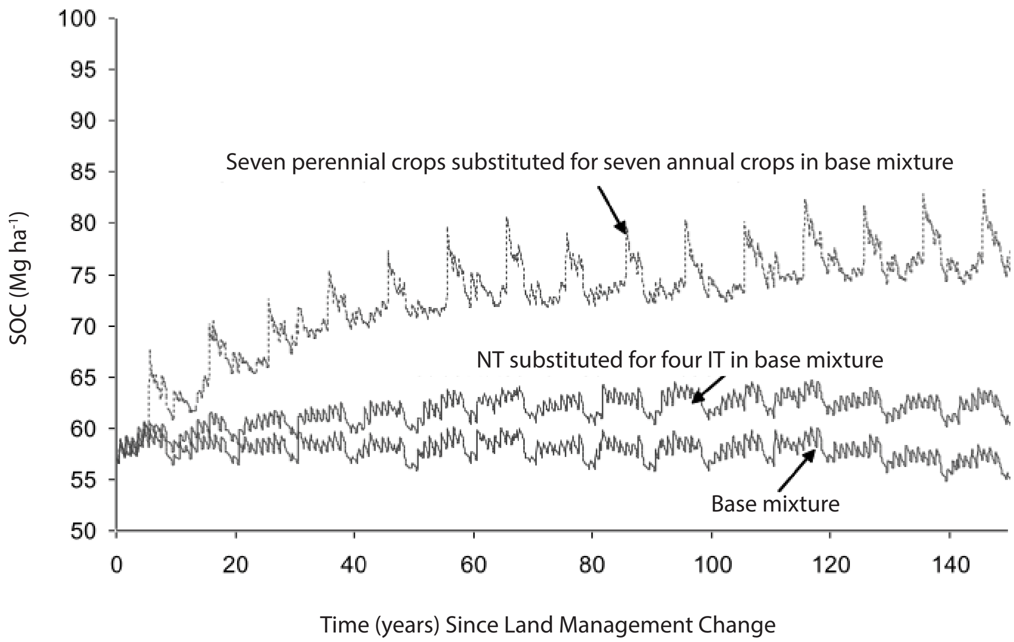


Notes:

a. Cumulative C gain over time

CTS = crop and tillage system

Figure A3.5–10 **Soil Organic Carbon (SOC) for a Base Crop Mix, for a Perennial (Alfalfa) Crop Substituted for an Annual Crop (Wheat) and for No-Till (NT) Substituted for Intensive Till (IT), in a Lethbridge Loam, Based on Century Runs**



Equation A3.5–2

$$\Delta C_{LMC}(t) = \frac{\Delta C}{P_{LMC}}$$

$\Delta C_{LMC}(t)$	=	change in SOC between land management systems in year *t* (Mg C ha^{-1})
ΔC	=	change in SOC due to LMC (Mg C)
P_{LMC}	=	proportion of the land area under a given land management system subject to the LMC, ha

This proportion (P_{LMC}) can be derived as the proportion of the particular land management system in the base system less the area under new land management system.

Equation A3.5–3

$$P_{LMC} = P_{LMbase} - P_{LMnew}$$

P_{LMC}	=	proportion of the land area under a given land management system subject to the LMC
P_{LMbase}	=	fraction of land management system of interest in the base land management system
P_{LMnew}	=	fraction of land management system of interest in the new land management system

Figure A3.5–10 provides an examples of Century runs for a Lethbridge loam (Orthic Dark Brown Chernozem) in the Semi-arid Prairies reporting zone. A base model run was performed using a 10-year base mix of crops based on the 1996 *Census of Agriculture* and weather data covering the 1951–2000 period. Century simulations of SOC were run by substituting perennial crops for the seven annual crops out of 10 in the base mixture. As a separate exercise, NT was substituted for IT for four years out of 10 in the base mixture (Figure A3.5–10). The next step was to calculate the $\Delta C_{LMC}(t)$ function by subtracting the simulated SOC values for the base mixture values from those imposed by the LMC of interest (Equation A3.5–2). Finally, the $\Delta C_{LMC}(t)$ was calculated as the proportion of the area of farming system divided by the P_{LMC}. In the case of the time series of ΔC_{LMC}, the respective values of P_{LMC} for the IT to NT reduction and for the addition of perennial crops were 4/10 and 7/10 (Figure A3.5–11).

SOC dynamics are believed to be governed by first-order kinetics, and thus the C change can be expressed as:

Equation A3.5–4

$$\Delta C_{LMC}(t) = \Delta C_{LMCmax} \times [1 - \exp(-kt)]$$

$\Delta C_{LMC}(t)$	=	change in SOC due to LMC at time t (Mg C ha^{-1})
ΔC_{LMCmax}	=	maximum SOC change induced by LMC (Mg C ha^{-1})
k	=	rate constant, yr^{-1}
t	=	year after LMC

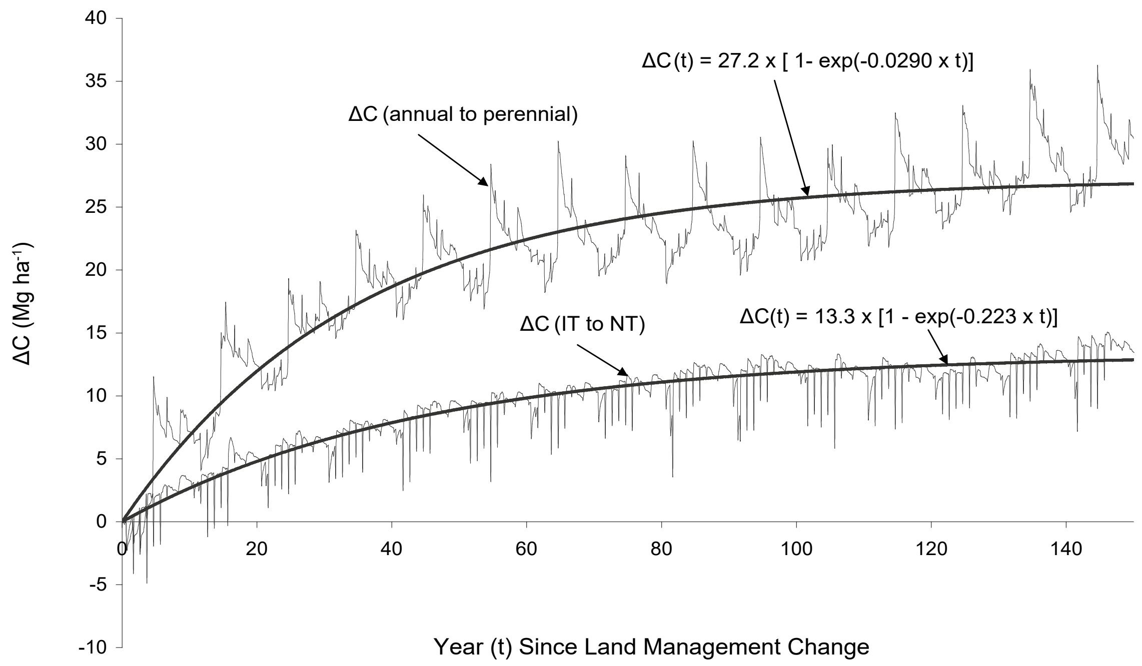
In practice, the exponential equations are fit statistically using a least squares method. The rates of change over time in Mg C ha^{-1} y^{-1} were established using a natural log transformation. These rates represent the instantaneous factor values for the LMC. Since the estimation is based on annual changes, the equation used for estimating the factor for annual change from the previous year (i.e. from year $t-1$ to year t) is:

Equation A3.5–5

$$F_{LMC}(t) = \Delta C_{LMCmax} \times [\exp\{-k \times (t-1)\} - \exp(-k \times t)]$$

$F_{LMC}(t)$	=	instantaneous C factor value due to LMC at time t, Mg C ha^{-1} yr^{-1}
ΔC_{LMCmax}	=	maximum SOC change induced by LMC (Mg C ha^{-1})
k	=	rate constant, yr^{-1}
t	=	year after LMC

Figure A3.5–11 **Change in SOC for Simulations with Substitutions Relative to Simulations with Base Crop Mix**



Since perfect steady-state conditions are never reached, the exponential equation should theoretically apply forever. In practice, however, the exponential equation was truncated when the $F_{LMC}(t)$ dropped to 25 kg C ha^{-1} yr^{-1}. This rate was below the practical measurement limit (Figure A3.5–12).

Estimating Mean k and ΔC_{LMCmax} for Practical Factor Calculations

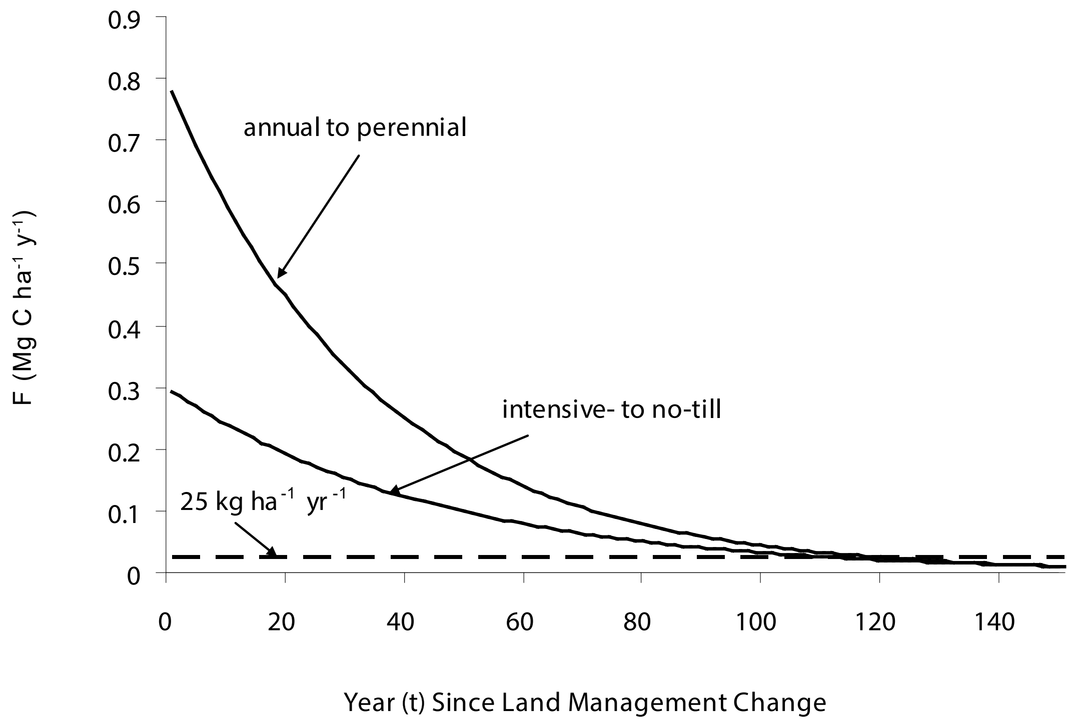
The ΔC_{LMCmax} and *k* parameters were determined for all 11 602 soil components in the CanSIS database and three LMCs (changes in tillage practices, summerfallow and annual-perennial crop mix). These soil components represent a wide range of initial SOC states and combinations of base crop mixtures and substitution amounts. The parameter values were estimated for each reporting zone as the mean of these soil components, weighted by the area of cropland on each component. The geometric mean was used for *k*, since its distribution was positively skewed. These means were calculated for three general soil textural classes (sandy, loamy and clayey) and applied to each soil component based on its textural class. Occasionally, *k* values of less than 0 resulted from the fit to ΔC_{LMC}; the *k* and ΔC_{LMCmax} from these fits were excluded from the reporting zone means.

Generally, the rates of SOC losses following an LMC are expected to be greater than the rates of SOC gains following the reverse LMC. However, this effect is highly dependent on the relative SOC amount at the time of the LMC. Documenting the SOC at the time of all LMCs is currently impossible. Consequently, to ensure transparency and simplicity, the reversibility assumption was imposed, whereby the SOC effect of an LMC in one direction is exactly the negative of the SOC effect of the same practice in the opposite direction.

Area-Based Soil Carbon Factor Validation – Tillage and Crop Mixtures

SOC change factors for LMCs used in the inventory were compared with empirical coefficients in VandenBygaart et al. (2008). This showed that the empirical data comparing the SOC change between IT and NT were highly variable, particularly for Eastern Canada. Nonetheless, the modelled factors were still within the range derived from the empirical data. The mean IT-NT factor derived from experiments in the Sub-humid Prairies reporting zone was over four times the value for the Semi-arid Prairies reporting zone. The mean Century model-derived factor for the Semi-arid Prairies reporting zone was similar to the factor derived from the field experiments. However, the Century-derived IT-NT factor for the Subhumid Prairies reporting zone was about 30% lower than the factor derived from the field experiments.

Figure A3.5–12 **Carbon Factors as a Function of Time**



The mean empirical factor for the switch from annual to perennial cropping was 0.59 Mg C ha^{-1} per year, which compares favourably with the range of 0.46–0.56 Mg C ha^{-1} per year in the modelled factors for the Parkland, Semi-arid Prairies and West zones (Table A6.5–4). In Eastern Canada, only two empirical change factors were available for the East Central zone, but they appeared to be in line with the modelled values (0.60–1.07 Mg C ha^{-1} per year empirical versus 0.74–0.77 Mg C ha^{-1} per year modelled).

Crop Productivity / Crop Residue C Inputs

Grain yields of major field crops in Canada have increased steadily since 1970 (Figure A3.5–13). The accurate estimation of crop productivity and quantification of crop residue C input on SOC storage at a regional or national scale over time rely on the use of spatio-temporal C input data. Estimating crop productivity / crop residue C input requires information on net primary productivity (NPP) and the proportion of NPP returned to the soil (Bolinder et al., 2007). NPP, measured as above-ground and below-ground plant biomass, is calculated using crop yield information. Fan et al. (2017) analyzed the harvest index and crop yield for 11 major crops from published field studies in temperate regions, and found significant linear relationships between harvest index and crop yield for wheat, maize, oats, barley, peas, chickpeas, lentils, soybeans, canola, and flax (r^2 = 0.19–0.65). These relationships between harvest index and crop yield were then used to develop estimates of crop residue inputs. For other crops, crop residue C is estimated using allocation ratios for grain, straw and roots from Janzen et al. (2003). The inclusion of soil C change estimates as impacted by crop productivity / crop residue C input is fully aligned with crop residue nitrogen (N) estimates that result in soil N_2O emissions (See Annex A3.4.5).

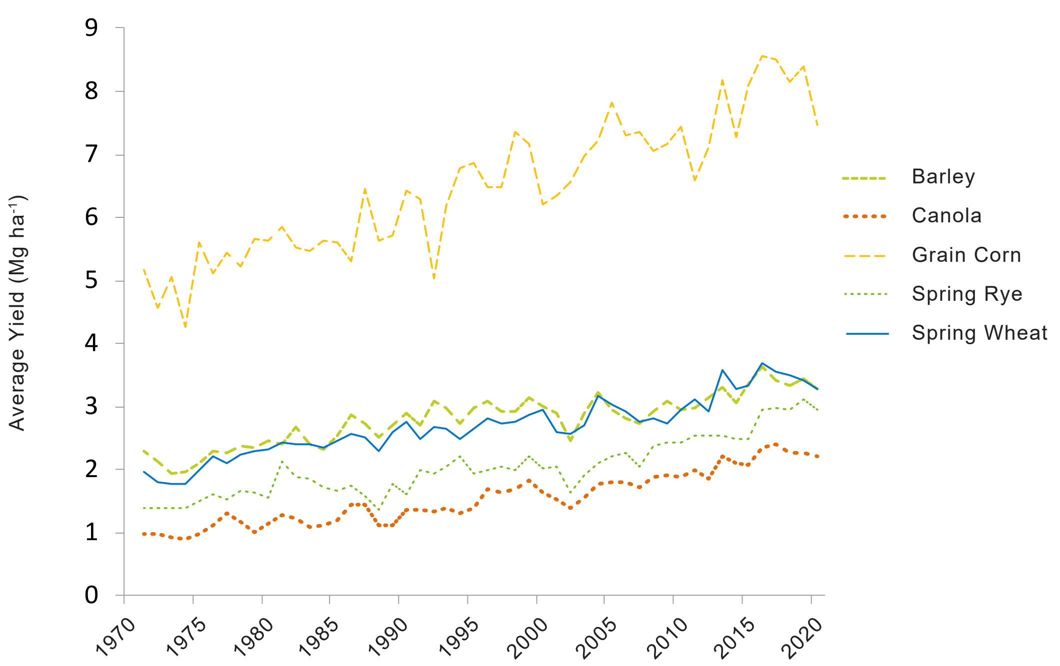
Crop residue C input is estimated using annual crop yield, taking into account the removal of crop residues by field burning and baling for each field crop as follows:

Equation A3.5–6

$$CRC_i = \sum(CRC\_AGR_{TC,i} + CRC\_BGR_{TC,i})/T\_AREA_i$$

CRC_i	=	crop residue C input estimated for all annual crops in ecodistrict *i*, from 1976 to 2020, Mg C ha^{-1} yr^{-1}
$CRC\_AGR_{TC,i}$	=	total above-ground (AGR) crop residue C from crop type (TC) in ecodistrict *i*, Mg C
$CRC\_BGR_{TC,i}$	=	total below-ground (BGR) crop residue C from TC in ecodistrict *i*, Mg C
T_AREA_i	=	total area of annual crops, including summerfallow, in ecodistrict *i*, ha

Figure A3.5–13 **Average Yields of Major Field Crops Grown in Canada from 1970 to 2020**



Note:
Data source: Statistics Canada, 2020, Table 32-10-0359-01

For major field crops, the harvest index is estimated by using a linear relationship with grain yield as reported by Fan et al. (2017).

Equation A3.5–7

$$\boldsymbol{HI_{TC} = YIELD_{TC,i} \times \left(1 - \frac{MC_{TC}}{100}\right) \times SLOPE_{TC} + INTERCEPT_{TC}}$$

HI_{TC}	=	harvest index for crop type (TC), %
$YIELD_{TC,i}$	=	grain yield for TC in ecodistrict *i*, Mg ha^{-1}
MC_{TC}	=	moisture content of grain yield for TC, %
$SLOPE_{TC}$	=	linear regression coefficient for TC as specified in Fan et al. (2017)
$INTERCEPT_{TC}$	=	intercept for TC as specified in Fan et al. (2017)

The dry matter portions of the above-ground residues and roots of major field crops are estimated using the harvest index and root-shoot ratio (Fan et al., 2017). For minor field crops, the dry matter portions of grain, above-ground residue and roots are calculated using the method proposed by Janzen et al. (2003).

Equation A3.5–8

$$\boldsymbol{CRC\_AGR_{TC,i} = \left[\left\{\sum PROD_{TC,i} \times \frac{PROD_{p,TC}}{\sum_{TC}(YIELD_{TC,i} \times AREA_{TC,i})}\right\} \times \left(1 - \frac{MC_{TC}}{100}\right) \times \frac{100}{HI_{TC,i}} \times \{1 - (BALE_{TC} \times 0.5 + BURN_{TC})\}\right] \times 0.45}$$

$$\boldsymbol{CRC\_BGR_{TC,i} = \left[\left\{\sum PROD_{TC,i} \times \frac{PROD_{p,TC}}{\sum_{TC}(YIELD_{TC,i} \times AREA_{TC,i})}\right\} \times \left(1 - \frac{MC_{TC}}{100}\right) \times \frac{100}{HI_{TC,i}} \times RSR_{TC}\right] \times 0.45}$$

$CRC\_AGR_{TC,i}$	=	above-ground crop residue C for crop type (TC) in ecodistrict *i*, Mg C
$PROD_{TC,i}$	=	total grain production for TC in ecodistrict *i*, Mg
$PROD_{p,TC}$	=	total grain production for TC in province *p*, Mg
$\sum_{TC}(YIELD_{TC,i} \times AREA_{TC,i})$	=	estimated grain production using average yield and seeded area for TC in ecodistrict *i*, Mg
MC_{TC}	=	moisture content of grain for TC, %
$HI_{TC,i}$	=	harvest index for TC in ecodistrict *i*, %
$BALE_{TC}$	=	baling of crop residues for TC, assuming 50% baling efficiency, fraction
$BURN_{TC}$	=	field burning of crop residue for TC, fraction
RSR_{TC}	=	root to above-ground crop residue ratio for TC, fraction
0.45	=	fraction of C in crop residue

SOC Change Factors

The 2019 Refinement to the 2006 IPCC Guidelines (IPCC, 2019) provides a Tier 2 steady state approach for estimating the change in SOC as impacted by crop productivity on land in the Cropland Remaining Cropland subcategory (IPCC, 2019). This approach considers three conceptual C pools (active, slow, and passive) in the top 30 cm of the soil profile. Decomposition is driven by decay rates that depend on temperature and soil moisture. Soil texture and tillage practices alter the decomposition of the active and slow C pools. Ogle et al. (2012) provide a more detailed discussion of this model. The approach used in the IPCC Tier 2 steady state model requires the lignin and N content of the C inputs (IPCC, 2019). The model is initialized using the first 10-year C inputs as recommended in the IPCC guidelines (IPCC, 2019). The initial average soil C stocks for model initialization were extracted from the National Soil Database[9] (NSDB, Version 3.2) for the dominant soil series in each textural class by ecodistrict. Figure A3.5–14 provides a schematic representation of the procedures for estimating soil C changes as impacted by crop productivity / crop residue C input.

9 Available online at: https://sis.agr.gc.ca/cansis/nsdb/index.html

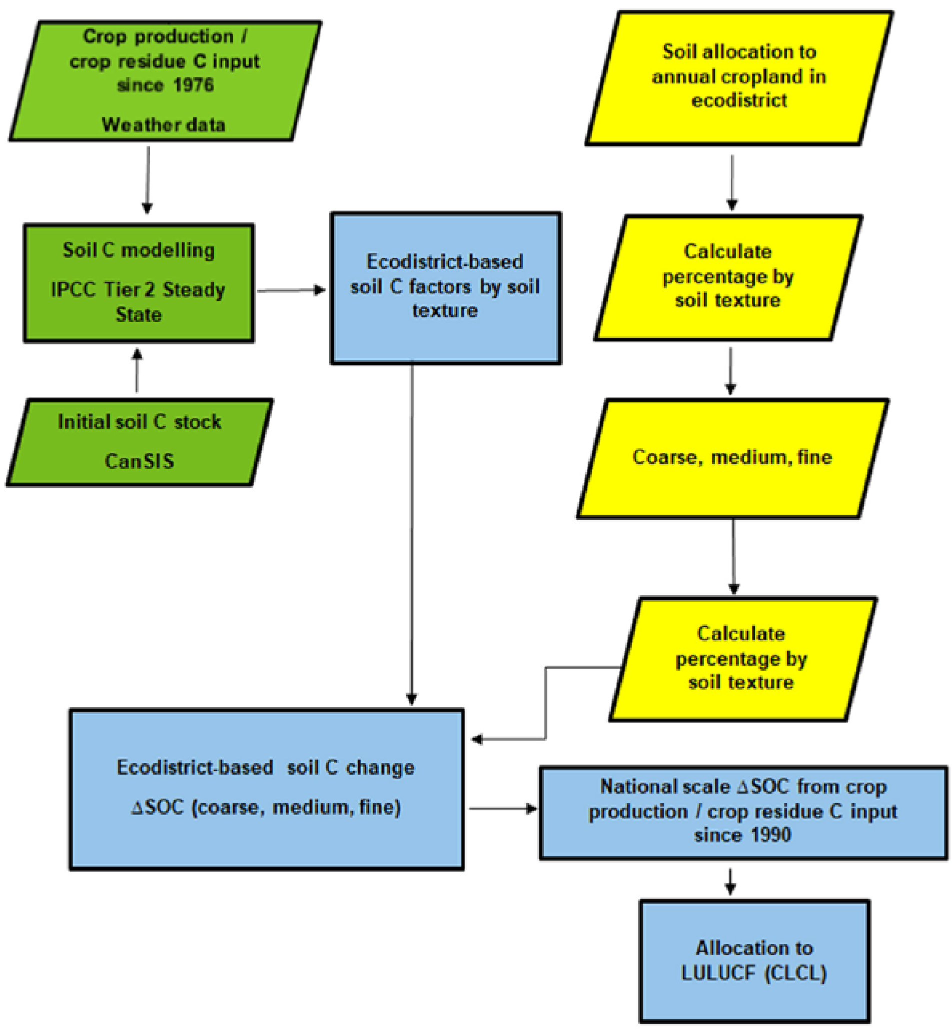
The IPCC Tier 2 Steady State approach was applied to Canadian soils to calculate a factor used to estimate the change in soil C associated with crop productivity. The change in SOC is calculated by applying a ΔSOC factor weighted by the proportions of textural classes in each ecodistrict to the total area in the ecodistrict for each consecutive year from 1976 to current inventory year. These estimates were combined with the LMC factors for tillage and annual/perennial crop mixtures. The total C change in an ecodistrict is then generated by summing SOC change values across all the textural classes present in the ecodistrict (Equation A3.5–9).

Equation A3.5–9

$$\Delta SOC_{t,i} = \sum\{(SOC_{t,TX,i} - SOC_{t-1,TX,i}) \times TF_{t,i,TX}\} \times T\_AREA_{t,i}$$

$\Delta SOC_{t,i}$	=	change in soil organic carbon (SOC) in ecodistrict *i*, and year *t*, Mg C
$SOC_{t,TX,i}$	=	SOC stock associated with soil texture class (TX) in year *t* and ecodistrict *i*, Mg C ha^{-1}
$SOC_{t-1,TX,i}$	=	SOC stock associated with soil TX in year *t-1* and ecodistrict *i*, Mg C ha^{-1}
$TF_{t,TX,i}$	=	soil texture fraction in coarse, medium and fine, TX in ecodistrict *i* and year *t*, fraction
$T\_AREA_{t,i}$	=	total area of annual crops, including summerfallow, in ecodistrict *i* and year *t*, ha

Figure A3.5–14 **Schematic Representation of Procedures for Estimating Soil Organic Carbon Changes as Impacted by Crop Productivity / Crop Residue Carbon Input**



Note:
CLCL = Cropland Remaining Cropland

Owing to the lack of certainty surrounding the actual C stocks and to maintain consistency with the other C factors, the method only reports the change in C since 1990, consistent with the area-based factor approach. The period from 1976 to 1989 is considered the baseline reference, and the mean change in C stocks resulting from the change in crop productivity (ΔC_{CP}) values relative to the reference period are then added to the ΔC_{LMC} values.

Manure Application

The application of animal manure on agricultural soils can increase SOC storage, particularly soils used for annual crop production. The manure-induced C retention (MCR) factor represents the average fraction of C input from various manures that is retained in the soil. This factor was established based on the measurements taken in multiple long-term studies. A country-specific method using MCR values is proposed to estimate the change in the soil C sink resulting from the application of manure to cropland soils in Canada. Eight field studies using various types and rates of manure application on different crop rotations, with durations varying from 10 to 74 years, were used to quantify MCR values under different climatic conditions in Canada. Solid cattle and swine manure had an MCR value of 26%, whereas the MCR value for liquid manure, including that from swine and cattle, was much lower, at only 5%. Compared with stockpiled manures, composted manure had a higher MCR (36%) due to the additional stabilization of C during the composting process (Liang et al., 2021). The use of individual MCR values for different manure types provides a sound approach to quantifying changes in soil C storage resulting from manure applications at a regional or national scale. According to the study by Liang et al. (2021), this approach is more accurate for Canadian conditions than the IPCC default ratio factors, which use a qualitative measure of manure input (IPCC, 2006).

Other variables required for estimating soil C storage from manure application include livestock populations, manure N excretion, and manure N losses during storage and handling (ammonia volatilization and N leaching). These data are consistent with those used in the estimation of emissions reported in the Enteric Fermentation and Manure Management categories in the Agriculture sector (see Annex A3.4). The amount of C in the manure was estimated using C:N ratios compiled in the literature review (Table A6.5–5).

Equation A3.5–10

$$\Delta C_{ManureC} = \sum(ManureN_{T,AWMS,i} \times CN_{T,AWMS}) \times MCR_{T,AWMS}$$

$\Delta C_{ManureC}$	=	amount of soil C gain from manure application, kg C ha^{-1} yr^{-1}
$ManureN_{T,AWMS,i}$	=	amount of manure N applied in ecodistrict *i*, for livestock type (*T*) and animal waste management system (*AWMS*) on annual crop, kg N ha^{-1} yr^{-1}
$CN_{T,AWMS,i}$	=	C:N ratio of applied manure for *T* and *AWMS*, unitless, as specified in Table A6.5–5
$MCR_{T,AWMS}$	=	manure-induced C retention for *T* and *AWMS*, fraction, as specified in Liang et al. (2021)

Estimates of Change in Soil Carbon Stocks

SOC changes as a result of LMC were reported for all inventory years since 1990. Because the effect of an LMC declines over time, the time period when the change was deemed to have occurred is maintained for each LMC. The C change factor was multiplied by the area of the LMC and summed across the soil components to produce the estimated SOC change for the SLC polygon. The SLC polygon is the smallest georeferenced unit of SOC stocks and SOC stock changes calculated using an IPCC Tier 2 approach as follows:

Equation A3.5–11

$$\Delta C_{LMC} = \sum_{1990-n} \sum_{ALL\ Ecod} (\Delta C_{TILL} + \Delta C_{CP} + \Delta C_{CROPPING} + \Delta C_{ManureC})$$

ΔC_{LMC}	=	change in SOC stocks due to LMC for a specific year after 1990 until year *n* (latest inventory year)
$ALLEcod$	=	all ecodistricts that contain land under cropland management practices
ΔC_{TILL}	=	change in SOC stocks due to change in tillage practices in each ecodistrict, since each particular tillage change
ΔC_{CP}	=	change in SOC stocks due to the change in crop productivity / crop residue C in each ecodistrict
$\Delta C_{CROPPING}$	=	change in soil C stocks due to the change in annual and perennial crops in each SLC
$\Delta C_{ManureC}$	=	change in soil C stocks due to the change in manure application in each ecodistrict

Figure A3.5–15 provides a schematic diagram of C change estimation methods.

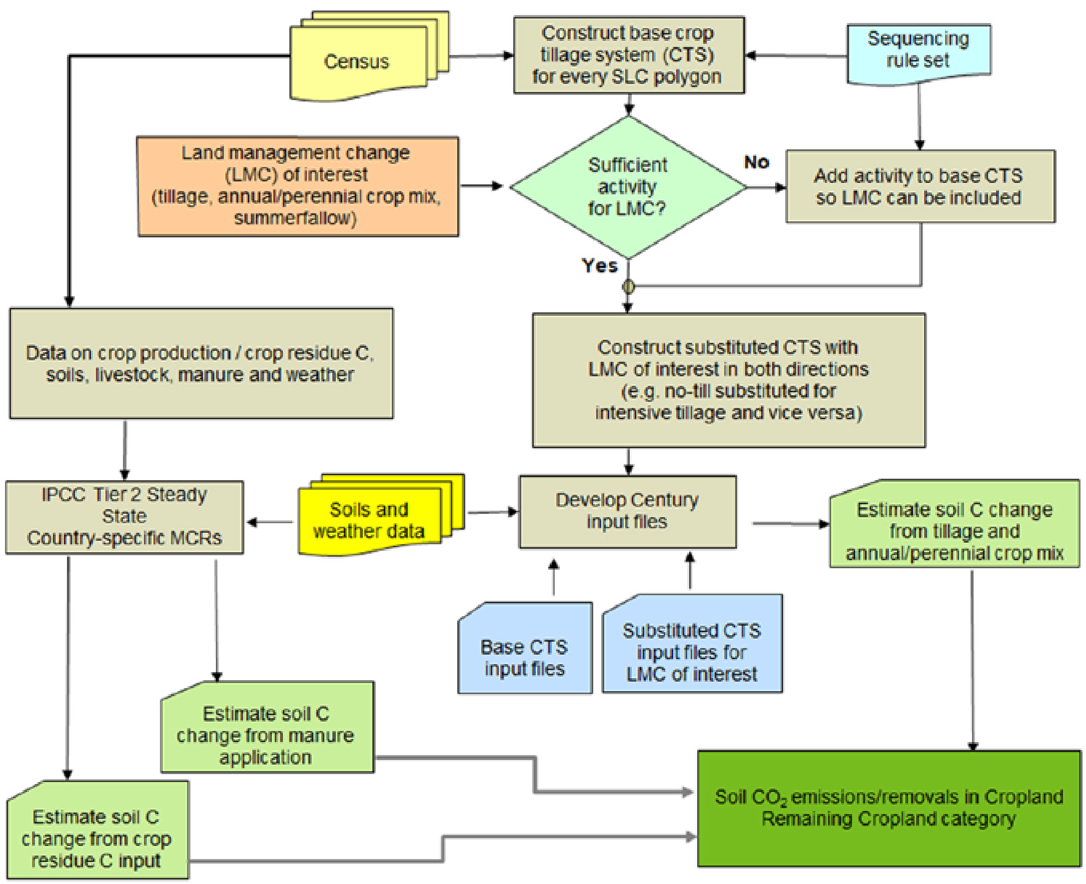
Data Sources

Carbon stock change estimates rely on C factors and a time series of land management data in the *Census of Agriculture*. Two types of data are used in deriving C factors (modelling) or calculating actual soil C stock change estimates. The main data used for modelling C factors include information from the SLC database, data on crop-tillage systems derived from the *Census of Agriculture*, crop yields, climatic data and activity data from other surveys and databases. Data on land management practices from the *Census of Agriculture* are the main inputs utilized in estimating annual soil C stock changes.

Land Information and Activity

The SLC is a national-scale spatial database describing the types of soils associated with landforms, displayed as polygons at a representation scale of 1:1 million.[10] The SLC was chosen for the LULUCF inventory because of its national scope and standardized structure, which ensure that all areas of the country are treated consistently with regard to inventory assessment procedures. The current version of the SLC in the National Soil Database (NSDB) data holdings is version 3.2; in this version, the coverage of soil attribute information is restricted primarily to the agricultural areas of Canada. In instances where agricultural land was mapped outside the coverage of SLC v3.2, soil attribute information was extracted from SLC version 2.2, which is older but covers all soils in Canada. All SLC polygons are nested within the 1995 National Ecological Framework, making it possible to scale up or scale down data and estimates, as required.

Figure A3.5–15 **Method of Using Area-Based and Input-Based Factors for Land Management Change to Estimate Carbon Change in Canada**



Note:
MCR = manure-induced C retention

10 Available online at http://sis.agr.gc.ca/cansis.

In all provinces within the Canada's agricultural ecumene, detailed soil survey information at map scales greater than 1:1 million was used to delineate the SLC polygons and compile the associated database files. The SLC Component Soil Names Tables and Soil Layer Tables provided specific input data, including soil C content, soil texture, pH, bulk density and soil hydraulic properties for modelling C factors with Century. The SLC polygon provides a spatial basis for allocating land management practices. *Census of Agriculture* provides information on tillage practices and cropping systems and estimates on the land area associated to cropland converted from forest and grassland—to modelled C factors. The estimated areas of cropland and other land-use practices in each SLC polygon were derived from EO-based maps for 1990, 2000, 2005, 2010, and 2015.

Analysis Units

There are 3475 SLC polygons in which agricultural activities occur. Since the SLC polygons have several soil landscape components, the finest spatial resolution for the analysis of agricultural activities in SLC polygons results in 13 771 unique combinations of soils, landforms and slope positions. These unique combinations represent the basic analysis units. The locations of land management types and soil components are not spatially explicit, but instead are spatially referenced to SLC polygons.

A procedure was developed to assign agricultural activities to the SLC polygons based on the suitability of each component of a soil polygon. The soil components have different inherent properties that make them more or less likely to be used for specific types of agricultural activities. Each soil component in the SLC attribute file has a suitability rating of high, moderate or low in terms of its likelihood of being under annual crop production. In this way, annual crop production is linked to those soils with a high rating. If the area with a high likelihood of being under annual cropland is insufficient to be linked to annual crop production, the remaining annual crop production will be assigned to components with a moderate likelihood of being under annual crop production and, if required, to low-ranked components. After the annual crop production area was assigned, perennial forages and seeded pasture area were linked to the remaining components in the same manner, starting with components with the highest likelihood of being under annual crop production and ending with components with the lowest likelihood of being under annual crop production.

Crop Yields

Crop yields at the ecodistrict level were developed from annual surveys conducted by Statistics Canada. Involving as many as 31 000 farmers, these surveys are stratified by region, and are used to compile estimates of the area, yield, production and stocks of the principal field crops grown in Canada. Several publications are released at strategic points in the crop year. Yields and levels of production by province are estimated twice, the first time in July based on expectations to the end of harvest and the second time, in November, after the harvest. The data are released at the small area data (SAD) region level, providing crop yields for approximately 70 spatial units in the country. SAD region boundaries were overlaid on SLC boundaries in a GIS, and a yield value for each crop in each soil polygon was assigned based on a majority rule. Data used to calibrate the Century crop growth sub-model included 1975–2004 yield data for wheat, barley, oats, corn, soybeans, potatoes and canola. Data used for crop C input modelling included all crop yield data since 1976. When crop yields were not available at the SAD region level, provincial yield statistics were used instead. These yield values were used to calibrate the Century crop growth sub-model for arriving at the area-based factors.

The amount of crop residue C that is returned to the soil at an ecodistrict scale since 1976 was quantified annually based on crop production data (Statistics Canada, 2020). Information on the baling of residues from various field crops is collected by Statistics Canada, through its Farm Environmental Management Survey (FEMS). Data on the field burning of crop residues are also available (see Annex 3.4.7). Crop residue inputs are consistent with crop residue calculations used in the computation of N_2O emissions in chapter 5 (Agricultural Soils).

Climatic Data

The database maintained by Agriculture and Agri-Food Canada (AAFC) contains 958 weather stations. Long-term normals for monthly maximum and minimum temperatures (T, ºC) and precipitation (mm) from 1951 to 2000 for all ecodistricts were used for the area-based modelling of C factors. For input-based models, the Environment and Climate Change Canada (ECCC) weather database (1981-2010) was used to ensure consistency with the climate parameters in the N estimation methodology. Application of the IPCC Tier 2 methodology also required Tmax, Tmin, Tavg, precipitation and potential evapotranspiration (PET) data. AAFC-archived weather data were provided by ECCC's Meteorological Service of Canada.

Earth Observation and the *Census of Agriculture*

Activity data for C stock estimation in the Cropland Remaining Cropland category rely mainly on a combination of data from the *Census of Agriculture* and area estimates based on EO analyses. The *Census of Agriculture* is conducted every five years to develop a statistical portrait of Canada's farms and agricultural operators. For confidentiality reasons, the smallest area for which Statistics Canada externally releases data from the *Census of Agriculture* is the dissemination/enumeration area level (of which there are approximately 52 000 in Canada). To provide a biophysical basis for modelling, data at this level were attributed to the SLC polygons (McConkey et al., 2007a).

EO-based mapping data were used to provide area estimates of all land-use practices within each of the agricultural SLC polygons in Canada. Land-use maps based on EO information were generated for 1990, 2000, 2005, 2010, and 2015 (Huffman et al., 2015a; AAFC, 2021). SLC polygons were used as the level of spatial stratification and data were compiled in seven primary land cover classifications: cropland, grassland, forest, settlements, wetlands, water, and other land cover. From 1990 to the latest inventory year, annual estimates of land-use areas were generated by interpolating between EO years and extrapolating beyond the last year of coverage. Agricultural land-use estimates prior to 1990 were obtained using *Census of Agriculture* data and relative changes in cropland and grassland areas between census periods. Land-use estimates for 1981 were calculated by determining the relative change in agricultural land use from 1991 and 1981 census data and applying this change to the 1990 EO data. Then, moving progressively back through the periods between census years, the relative changes were used to generate agricultural land-use estimates back to 1951. To minimize the spatial variability associated with known issues involving the reporting of land-use areas based on farm headquarters, the relative change in land-use estimates was calculated at the ecodistrict scale and applied to all nested SLC polygons.

The EO-based cropland attributes were estimated using ratios of cropland area attributes to total cropland area from the *Census of Agriculture*. To reduce the differences between the EO-based and census-based estimates of provincial crop areas, EO cropland categories (i.e. cropland, pasture, orchards and vineyards) were reconciled using provincial scaling factors. Reconciliations were constrained by the total area of agricultural land within SLC polygons, as interpreted through EO analysis. Data on tillage management practices were taken from the *Census of Agriculture* in the following categories: IT (tillage that incorporates most of the crop residue in the soil); RT (tillage that retains most of the crop residue on the surface); and NT (no-till seeding or zero-till seeding). For summerfallow, the following tillage categories were used: NT (the area on which chemicals were exclusively used for weed control); IT (the area on which tillage only was used); and RT (the area on which a combination of tillage and chemicals were used). More details on the methodological approach used to create EO-based agricultural activity data are provided in Cerkowniak (2019).

Uncertainty

The derivation of uncertainties associated with estimates of CO_2 emissions and removals requires estimates of uncertainties for LMC areas and the C factors associated with changes in tillage and annual/perennial crops (McConkey et al., 2007b). The uncertainty described in this report is based on the 2014 submission methodology and has not yet been updated for the new EO methodology.

The uncertainty surrounding the area of change was determined for ecodistricts. The average area of agricultural land within an ecodistrict is about 140 kha, i.e. sufficiently large that the areas of different management practices were considered independent of those in others, including adjacent ecodistricts. Errors in the areas of management practices in each ecodistrict were assumed to represent inherent uncertainty that was unaffected by the uncertainty of those in other ecodistricts. Furthermore, the ecodistrict area is sufficiently large that a null report for an activity can be assumed to mean that the activity is not occurring within the ecodistrict. Therefore, the uncertainty over area can be more for an ecodistrict than for an SLC polygon when considered in relative terms.

The uncertainty surrounding the area of a management practice in an average ecodistrict at a given time was based on the relative proportion of the area of that management practice in the ecodistrict. The relative uncertainty of the area of a management practice expressed as the standard deviation of an assumed normal population decreased from 10% of the area to 1.25% of the area as the relative area of that practice increased.[11]

The uncertainties associated with C change factors for fallow, tillage and annual/perennial crops were assumed to arise from two main sources: (1) the process uncertainty associated with C change due to inaccuracies in predicting C change even if the situation of the management practice was defined perfectly; and (2) the situational uncertainty associated with C change due to variation in the situation of the management practice.

Process uncertainty includes the effect of uncertainty in the model. This includes the uncertainty in the model predictions due to uncertain model parameters and the inaccurate and/or incomplete representation of all relevant processes by the model. When empirical data are used, process uncertainty includes inadequacies in measurement techniques, analysis

11 Huffman T. 2006. Personal communication (from Huffman T, Agriculture and Agri-Food Canada to McConkey BG, Agriculture and Agri-Food Canada).

error, poor representativeness of measurements, and failure to measure components of C change. To estimate the process error, the variation from measured C change values in controlled experiments was used. It was assumed that this represents the inherent uncertainty even when the situation is accurately described. Process uncertainty scaling coefficients for tillage and fallow were derived for Canada from VandenBygaart et al. (2003).

Situational uncertainty derives from the inability to accurately describe each situation. This includes the effect of interactions with past or concurrent changes to land use or land management and variability in the weather, soil properties, crop management and LMC continuity. The situational uncertainty scaling coefficients for fallow change, tillage change and annual perennial crop change were estimated based on the observed variability of Century-simulated C change for all soil component–management–climate combinations within the RU. C change was calculated for many management combinations. In addition, a range of historical ecodistrict weather data was included in the Century simulations. The situational uncertainty also includes the additional variability of the regional factors introduced by the imposition of C change reversibility. Average situational uncertainty scaling coefficients were derived for Canada (McConkey et al., 2007b).

Although process and situational uncertainty are expected to interact, describing their relationship is not feasible given the complexity resulting from the large number of possible interactions between deviations due to process uncertainty and to situational uncertainty. Hence, it was assumed that the total deviation in total C change was the sum of the deviations resulting from process and situational uncertainty. Details of uncertainty estimate development are provided in McConkey et al. (2007b). The results of this analysis are outlined in Chapter 6.5.1.

A formal uncertainty analysis has not yet been carried out for the estimates of cropland C change associated with changes in crop yield. However other soil C models of varying complexity (i.e. Rothamsted carbon model [RothC], Introductory Carbon Balance Model [ICBM)] and Campbell model) that are capable of using measured yields as C inputs in simulations were tested in the national C assessment analysis. These models were also used for simulations of SOC, with a varying degree of success relative to field observations (Thiagarajan et al., 2022). Estimates of national C change varied from a loss of 5.4 Mt C in 1990 to a gain of 4.4 Mt C in 2020 with the Campbell model, and gains of 9.9 Mt C and 38.5 Mt C in 1990 and 2020, respectively, with the ICBM model. To provide comparability among Annex 1 Parties, the IPCC Tier 2 approach is used for estimating the change in SOC as impacted by crop productivity / crop residue C input at the ecodistrict, RU and Canada-wide spatial scales. The results of this approach were observed to be roughly equivalent to the mean of the other models, suggesting that it provides estimates within the bounds of what has been typically observed in other studies, as represented by the default parameters in existing soil C models.

Similarly, a formal uncertainty analysis has not been conducted on the estimation of cropland C change resulting from manure application, although uncertainty estimates associated with field measurements of MCRs are available: ±30% for solid swine and cattle manure, ±13% for swine manure compost, and ±330% for liquid swine and cattle manure. Discussions of the uncertainties associated with livestock types, populations, and manure excretion as well as the C:N ratio in various types of manures can be found in sections A3.4.1, A3.4.3 and Table A6.5–5 respectively.

CO_2 Emissions and Removals from Woody Biomass

Estimates of emissions and removals from woody biomass on croplands include those originating from trees and shrubs on agricultural land as well as in vineyards, fruit orchards and Christmas tree plantations. A remote sensing-based sampling approach was used to determine areas of trees and shrubs during the reporting period, while the Census of Agriculture provided estimated areas of vineyards, fruit orchards and Christmas tree farms.

Vineyards, fruit orchards and Christmas tree farms are intensively managed for sustained yields. Vineyards are pruned each year, leaving only the trunk and one-year-old stems. Similarly, fruit trees are pruned annually to maintain the desired canopy shape and size. Old plants are replaced on a rotating basis for disease prevention, stock improvement or introduction of new varieties. Typically, Christmas trees are harvested at about 10 years of age. For all three crops, it was assumed that, because of these rotational practices and the requirements for sustained yield, a uniform age-class distribution is generally found on production farms. Hence, there would be no net increase or decrease in biomass C in existing farms, since the C lost from harvest or replacement would be balanced by gains due to new plant growth. The approach was therefore limited to detecting changes in areas under vineyards, fruit orchards and Christmas tree plantations and estimating the corresponding C stock changes in total biomass.

There are no Canadian studies on above-ground or below-ground C dynamics in vineyards or fruit orchards. However, results from other studies are considered valid, since the varieties, field production techniques and even root stocks are often the same. Canadian literature on Christmas tree plantations is used whenever suitable.

On average, vines are replaced at 28 years of age; the average vine is therefore 14 years old (Mailvaganam, 2002). Because of intensive pruning, linear rates of above-ground and below-ground biomass accumulation in trunks and roots were established at 0.4 and 0.3 Mg ha^{-1} yr^{-1}, respectively (Nendel and Kersebaum, 2004). These were converted to C values using a 50% C content in biomass. Upon a decrease in vineyard areas, an instantaneous loss of 4.9 Mg C ha^{-1} is assumed, equal to the average standing biomass for 14-year-old vines (McConkey et al., 2007a).

Because of different standard planting densities, the range of standing biomass per area for apple and peach orchards varied narrowly between 36 and 40 Mg ha^{-1} (McConkey et al., 2007a). This similarity is expected since, regardless of tree size and planting density, the tree shapes and canopies are manipulated to maximize net photosynthesis per area. The annual rate of C sequestration was calculated over a 10-year growth period at 1.6 Mg C ha^{-1} yr^{-1}. The same rate, multiplied by a root-to-shoot ratio of 0.4 (Bartelink, 1998), was used to estimate C sequestration in below-ground biomass. Instantaneous C loss upon a decrease in the area of orchards was equal to 50% of the total biomass of a 10-year-old tree (22.4 Mg C ha^{-1}).

Christmas trees are marketed at about 10 years of age (McConkey et al., 2007a). With a root-to-shoot ratio of 0.3 (Bartelink, 1998; Litton et al., 2003; Xiao and Ceulemans, 2004), the total C biomass of a marketable tree plantation is estimated at 11.1 Mg C ha^{-1}. Carbon sequestration in the biomass of new Christmas tree plantations is calculated at five years at rates of 0.85 and 0.26 Mg C ha^{-1} for above-ground and below-ground biomass, respectively. A decrease of plantation area would result in the immediate loss of 5.6 Mg C ha^{-1}.

Trees and shrubs on agricultural land include perennial woody cover types in farmyards, shelterbelts and hedgerows. Carbon storage in woody biomass in the landscape changes over time as trees and shrubs grow and die; areas of land with woody biomass change due to planting or colonization of cropland areas; or tree clearing takes place.

The EO-based sampling approach used to quantify changes in woody biomass on Canadian croplands was developed by Huffman et al. (2015b). Briefly, the National Ecological Framework (Marshall et al., 1999) was used to develop a spatially stratified, random sampling approach, with 30 sample sites per ecozone targeted as the objective. High-resolution historical aerial photos from the National Air Photo Library of Natural Resources Canada and from provincial databases were selected to digitize the land cover of tree and shrub in a 2 km by 2 km plot at circa 1990, circa 2000 and circa 2010 at a 1:10000 scale. The "trees" land cover class was defined as having less than 25% crown closure and occupying less than 1 ha. The "shrubs" land cover class represents non-agricultural woody plants that would not be expected to meet the forest or "trees" definition when mature. Wood volume yield estimates for each ecozone were derived based on published literature and consultations with provincial forestry and agriculture specialists, conservation associations and researchers in academia. Overall, estimates of above-ground wood volume ranged between 99.3 and 181.7 m^3 ha^{-1} across ecozones and estimates of mean annual increments, between 1.2 and 3.8 m^3 ha^{-1} yr^{-1}. With the addition of a new data set in 2020, the growth, loss and gain in the biomass of trees and shrubs (in tonnes of C) were calculated for two time periods: 1990-2000 and 2000-2010 in croplands. An interpolation splicing technique from the 2006 IPCC Guidelines is applied to the transition between the two time periods to improve smoothness of the time-series. The analysis, coefficients and parameters used to estimate C stock changes were based on the methodology described by Huffman et al. (2015b) for both time periods.

An analysis of firewood production suggested that agricultural lands serve as an important source of fuel for residential bioenergy production in Canada (Doyon et al., 2019). A portion of the tree biomass lost from cropland was therefore transferred to the HWP pool as firewood input to meet regional residential bioenergy requirements (see section A3.5.3) at the RU level. Furthermore, in regions with a shortage of forest biomass supplies in a given RU, fractions of lost tree biomass from cropland were assumed to be sourced from the neighbouring RU. To avoid the double counting of emissions in the Cropland and Harvested Wood Products categories, the amount of C transferred to the HWP pool was not treated as C loss under Cropland, although it normally would be reported as immediately oxidized. As a result, what appears to be a larger sink or lower emissions from woody biomass is reported; however, once the transfer to HWP is taken into account, no net change in total C emissions or removals occurs.

Uncertainty

Orchards and vineyards with poor growth are regularly removed and replaced, and fruit trees and vineyards are often irrigated to maintain desired growth during dry periods. Consequently, the variability in C stock changes should be less than that for other agricultural activities.

In the case of loss of area, all C in woody biomass is assumed to be immediately released. There are no Canadian-specific data on the uncertainty surrounding vineyards, orchards and Christmas tree plantations. Therefore, the default uncertainty from the 2006 IPCC Guidelines of ±75% for woody biomass on cropland was used for these land cover types. The error propagation approach described in Huffman et al. (2015b) was applied to trees and shrubs. Although the loss in area of fruit trees, vineyards or Christmas tree farms is estimated to have been replaced by annual crops, a perennial-to-annual crop conversion is also deemed to occur, with an associated C change uncertainty that contributes to the overall C change uncertainty for a reporting zone.

Cultivation of Organic Soils

The cultivation of histosols for annual crop production usually involves drainage, tillage and fertilization. All these practices increase SOC decomposition and thus the release of CO_2 to the atmosphere.

Methodology

The IPCC Tier 1 methodology is based on the rate of C released per unit land area:

Equation A3.5–12

$$C = \sum(A_i \times EF)$$

C	=	C emissions from cultivation of organic soils (Mg C yr^{-1})
A_i	=	area of organic soils that is cultivated for annual crop production in province *i*, ha
EF	=	C emission factor, Mg C loss ha^{-1} yr^{-1}. The default emission factor of 5.0 Mg C $ha^{-1}yr^{-1}$ was used (IPCC, 2006).

A
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Data Sources

The *Census of Agriculture* does not provide information on areas of cultivated histosols by province. In the absence of these data, consultations were undertaken with numerous soil and crop specialists across Canada. On the basis of these consultations, the total area of cultivated organic soils in Canada was estimated to be 16 kha (Liang et al., 2004).

Uncertainty

The uncertainty associated with emissions from this source is due to the uncertainties surrounding the emission factor and the area estimates for cultivated histosols. A ±50% uncertainty level has been assigned to the 95% confidence limits associated with the area estimate of cultivated histosols, while a ±90% uncertainty level has been assigned to the 95% confidence limits for the emission factor provided in the 2006 IPCC Guidelines (IPCC, 2006).

A3.5.4.2. Grassland Converted to Cropland

The conversion of native grassland to cropland results in losses of SOC and soil organic nitrogen (SON) and in turn leads to emissions of CO_2 and N_2O to the atmosphere. According to a study on the burning of managed grasslands in Canada by Bailey and Liang (2013), C changes from above-ground or below-ground biomass or DOM upon conversion are generally insignificant. The authors reported that the average above-ground biomass was 1100 kg ha^{-1} on Brown Chernozem soils and 1700 kg ha^{-1} on Dark Brown Chernozem soils. The above-ground biomass for managed grassland would be lower than its yield under crop production (Liang et al., 2005).

A number of studies on changes in SOC and SON in grassland converted to cropland have been carried out in the Brown, Dark Brown and Black soil zones of the Canadian Prairies, and their results are summarized by McConkey et al. (2007a).

Losses of Soil Organic Carbon

The average loss of SOC based on field observations was 22% (McConkey et al., 2007a). Many of the studies involved comparisons within 30 years of breaking of the native grassland, whereas others were 70 or more years from breaking. Since many of these studies did not specify the period since breaking, it is assumed that the 22% SOC loss would refer to about 50–60 years after the land was broken.

The SOC dynamics from conversion of grassland to cropland on Brown and Dark Brown Chernozemic soils (Figure A3.5–16) can be estimated with the Century model (Version 4.0). Shortly after breaking, there is an increase in soil organic matter, as the below-ground biomass of the grass becomes part of the SOC. After a few years, the SOC declines to less than the amount of SOC that existed under grassland. The rate of SOC decline gradually decreases with time. When the initial SOC increase due to the added C from roots is excluded, the simulated SOC dynamics can be described by the following equation:

Equation A3.5–13

$$\Delta C(t) = \Delta C_{Bmax} \times [1 - exp\{-k \times (t - t_{lag})\}]$$

$\Delta C(t)$	=	change in SOC for the t^{th} year after conversion, Mg C ha^{-1}
ΔC_{Bmax}	=	ultimate change in SOC from grassland to cropland, Mg C ha^{-1}
k	=	rate constant for describing decomposition, yr^{-1}
t	=	time since breaking of grassland, years
t_{lag}	=	time lag before ΔC becomes negative, years

Assuming that the 22% loss at about 50–60 years after initial breaking represents the total loss, the ΔC_{Bmax} is 0.22/(1−0.22), or 28% of the stabilized SOC under agricultural land use. Given the uncertainty associated with the actual dynamics, no time lag was assumed to occur in SOC loss from breaking grassland, so that SOC starts to decline immediately upon breaking. With these assumptions, the general equation for predicting SOC loss from breaking grassland becomes:

Equation A3.5–14

$$\Delta C(t) = 0.28 \times SOC_{agric} \times [1 - exp(-0.12 \times t)]$$

$\Delta C(t)$	=	change in SOC for the t^{th} year after conversion, Mg C ha^{-1}
t	=	time since breaking, years
SOC_{agric}	=	0- to 30-cm SOC from the National Soil Database within CanSIS under an agricultural land use (Cropland category), Mg C ha^{-1}

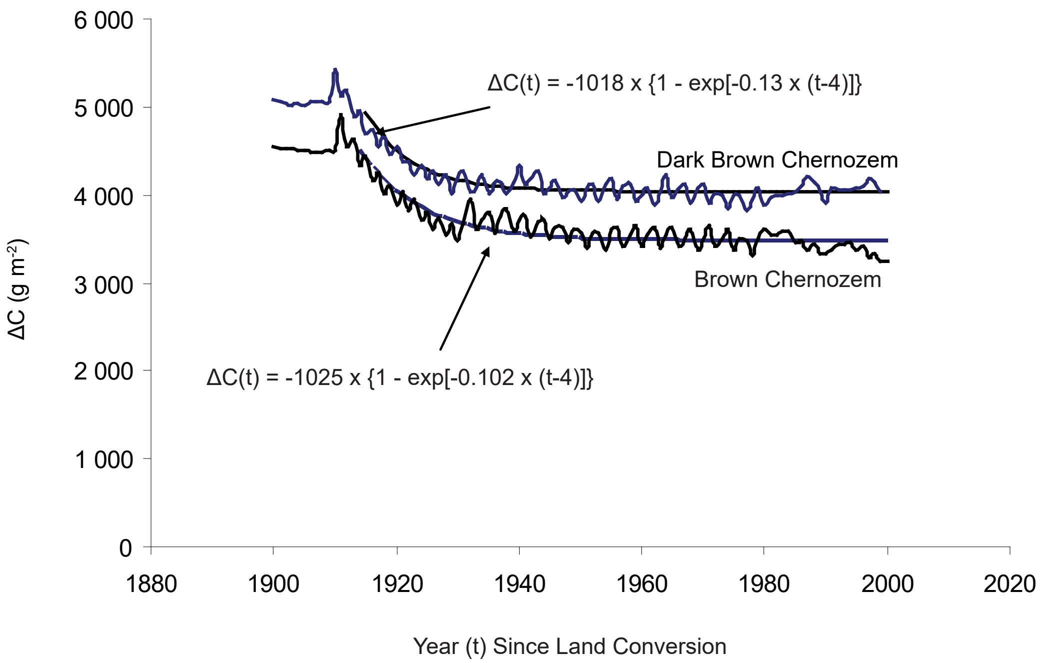
Thus, the total losses of SOC in grassland converted to cropland were calculated using an IPCC Tier 2 approach:

Equation A3.5–15

$$\Delta C_{GLCL} = \sum_{1951-n} \sum_{ALL\ SLC} \sum_{t} (\Delta C_t \times AREA_{GLCL})$$

ΔC_{GLCL}	=	losses of SOC in the inventory year *n* due to conversion of grassland to cropland since 1951 until year *n*, Mg C
$ALLSLC$	=	all soil polygons that contain grassland conversion to cropland
t	=	time after grassland conversion, years
ΔC_t	=	change in SOC for the t^{th} year after conversion, Mg C ha^{-1}
$AREA_{GLCL}$	=	area of grassland converted to cropland annually since 1951, ha

Figure A3.5–16 **SOC Dynamics after the Breaking of Grassland to Cropland on Brown and Dark Brown Chernozemic Soils, Simulated with the Century Model**



Losses of Soil Organic N and N_2O Emissions

The change in SON is estimated as a fixed proportion of C losses. Where changes in both SON and SOC were determined, the average change in SON was 0.06 kg N lost/kg C lost (McConkey et al., 2007a). Consequently, N_2O emissions from the conversion of grassland to cropland were calculated using an IPCC Tier 2 approach:

Equation A3.5–16

$$N_2O_{GLCL} = \sum_{1951-n}\sum_{ALLSLC}\sum_{t}(\Delta C_{GLCL} \times AREA_{GLCL}) \times 0.06 \times \mathrm{EF\_Base} \times \mathrm{RF\_SN} \times \mathrm{RF\_TX} \times \mathrm{RF\_NSE} \times \frac{44}{28}$$

N_2O_{GLCL}	=	emissions of N_2O in year *n* due to the conversion of grassland to cropland from 1951 to year *n*, kt
ALLSLC	=	all soil polygons containing grassland conversion to cropland
t	=	time after grassland conversion, years
ΔC_{GLCL}	=	change in SOC for the t^{th} year after grassland conversion, Mg C ha^{-1}
$AREA_{GLCL}$	=	area of grassland converted to cropland annually since 1951, ha
EF_Base		N_2O emission factor, defined as a function of long-term climate normals (growing season precipitation from May to October at the ecodistrict level) (see section A3.4.5)
RF_SN	=	ratio factor for adjusting the effect of source of N in soil N_2O emissions (see section A3.4.5)
RF_TX	=	ratio factor for adjusting soil texture (TX) effect on soil N_2O emissions (see section A3.4.5)
RF_NSE	=	ratio factor for correcting non-growing season soil N_2O emissions (see section A3.4.5)
0.06		ratio of SON to SOC losses
44/28		coefficient converting N_2O-N to N_2O

Data Sources

The area of grassland reported in the Grassland Remaining Grassland category was estimated using a combination of *Census of Agriculture* and EO data. Area estimates reported in the Grassland Converted to Cropland category were based on the reconciliation of changes in land area between Grassland Remaining Grassland and land under cropland management. To avoid issues associated with farm headquarters reporting, data were aggregated to the ecodistrict level prior to the land reconciliation process. Estimates of areas of grassland converted to cropland at the ecodistrict level were then apportioned back to SLC polygons.

Within each SLC polygon, areas under Grassland Remaining Grassland were allocated to soil components identified as having a "low" likelihood of being cropped. Soil C data from the National Soil Database were used to calculate an average SOC content for soils in each SLC polygon.

Uncertainty

The conversion of agricultural grassland to cropland occurs, but the reverse does not. The uncertainty associated with the converted area in a given ecodistrict cannot be greater than the uncertainty associated with the final area of cropland or the initial area of grassland. Therefore, the uncertainty associated with the conversion area was assumed to be equal to the uncertainty over the area of land in the Cropland or Grassland category, whichever was lower. The factor scaling coefficient was assumed to be the same as for annual-perennial crop conversions (McConkey et al., 2007b).

A3.5.4.3. **Forest Land Converted to Cropland**

Emissions of CO_2 and N_2O from Soils

The clearing of forest to increase the area of agricultural land is a declining but still significant practice in Canada. This section describes the methodology for estimating CO_2 and N_2O emissions associated with soil disturbance, while the methodology for estimating emissions from biomass after conversion is presented in sections A3.5.2.1 and A3.5.2.10. For SOC changes, it is necessary to differentiate between Eastern and Western Canada.

Eastern Canada

A number of studies have compared SOC values for forested land with SOC values for adjacent land under agriculture in Eastern Canada. The mean C loss was 20.3% for an approximately 0-30 cm depth (McConkey et al., 2007a). This value is comparable to that found in the CanSIS soil database (Table A6.5–9), indicating that, on average, SOC in the uppermost 30 cm of soil under agriculture was 20.5% less than that in soil under forest.

Although SOC values for forested land includes the C in the litter layer above the mineral soil, in practice, uncertainty is always involved in quantifying this source of C, as well as the organic C in soil debris (Paul et al., 2002). Soil erosion, which is generally assumed to increase under agriculture, also reduces measured SOC values on agricultural land.

The Century model (version 4.0) was used to estimate the SOC dynamics from forest conversion (Figure A3.5–17). In the first years after conversion, soil organic matter increases as litter and above-ground and below-ground DOM become part of SOC. After a few years, SOC falls below the pre-conversion value that existed before forest conversion. The rate of SOC decline gradually decreases with time.

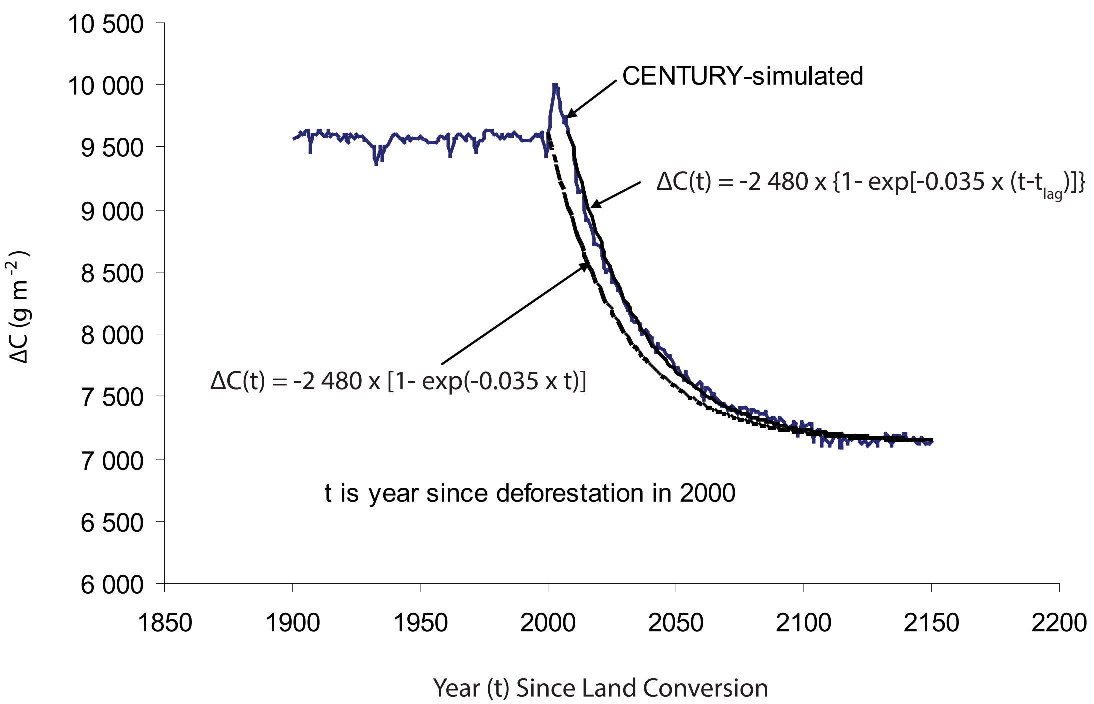
The following equation was fit to the Century results in Figure A3.5–16, excluding the initial SOC increase:

Equation A3.5–17

$$(\Delta C)_{(t)} = \Delta C_{Dmax} \times [1 - exp\{-k \times (t - t_{lag})\}]$$

$\Delta C(t)$	=	change in SOC for the t^{th} year after conversion, Mg C ha^{-1}
ΔC_{Dmax}	=	maximum change in SOC from forest conversion to agriculture, Mg C ha^{-1}
k	=	rate constant describing decomposition, yr^{-1}
t	=	time since conversion of forest land, years
t_{lag}	=	time lag before ΔC becomes negative, years

Figure A3.5–17 **Soil Organic Carbon Following Conversion of Deciduous Forest to Cropland, as Simulated by the Century Model**



In simulations of SOC after the conversion of deciduous forest to cropland (Figure A3.5–18), 25% of C losses occur within 20 years of forest conversion and 90%, within 100 years. Given the uncertainty in the actual dynamics, it was assumed that there is no time lag in SOC loss from forest conversion, so that SOC starts to decline immediately upon conversion, i.e. the fitted SOC loss (Figure A3.5–13) is used to estimate SOC loss with the time lag set to 0 after fitting.

The mean SOC loss of 20.5% resulting from forest conversion to cropland in Eastern Canada, based on CanSIS information, was assumed to correspond to the value approximately 100 years after forest conversion. The ΔC_{Dmax} is therefore corrected by a factor of 1/0.927, which assumes that only 92.7% of the C has been lost after 100 years, based on the integration of Equation A3.5–18, resulting in a ΔC_{Dmax} value of 22.1% of SOC under long-term forest. Since the CanSIS soil database has more data on SOC under long-term cropland conditions than on SOC under long-term forest conditions in areas where cropland exists, the maximal SOC losses were calculated relative to stabilized cropland SOC (i.e. loss = 0.221/(1−0.221) × SOC or loss = 0.284 × SOC under agriculture). Therefore, the final equation for estimating SOC loss from forest conversion to cropland in Eastern Canada is:

Equation A3.5–18

$$\Delta C(t) = 0.284 \times SOC_{agric} \times [1 - \exp(-0.0262 \times t)]$$

$\Delta C(t)$	=	change in SOC for the t^{th} year after conversion, Mg C ha^{-1}
SOC_{agric}	=	0- to 30-cm SOC for a cropland soil, Mg C ha^{-1}, according to CanSIS
-0.0262	=	rate constant for describing decomposition, yr^{-1}
t	=	time since conversion, years

Thus, the total amount of SOC lost from forest land converted to cropland can be estimated using the following equation:

Equation A3.5–19

$$\Delta C_{FLCL} = \sum_{1970-n} \sum_{ALLSLC} \sum_{t} (\Delta C_t \times AREA_{FLCL})$$

ΔC_{FLCL}	=	total SOC loss in year *n* from the conversion of forest land to cropland from 1970 to year *n*, Mg C ha^{-1}
t	=	time after conversion, year
$ALLSLC$	=	all soil polygons that contain forest land converted to cropland
ΔC_t	=	change in SOC for the t^{th} year after conversion, Mg C ha^{-1} (see Equation A3.5–18)
$AREA_{FLCL}$	=	area of forest land converted to cropland annually since 1970, ha

Note that the SOC loss predicted by Equation A3.5–19 is in addition to C stock changes in the tree biomass and woody DOM present in the forest at the time of forest conversion.

According to field observations, the average N change in Eastern Canada was -5.2%, representing 0.4 Mg N ha^{-1} (McConkey et al., 2007a). For those comparisons where both N and C losses were determined, the corresponding C loss was 19.9 Mg C ha^{-1}, and the C loss was 50 times the N loss. For simplicity, it was assumed that the N loss was a constant 2% of the C loss. Thus, N_2O emissions from the conversion of forest land to cropland are estimated using the following equation:

Equation A3.5–20

$$N_2O_{FLCL} = \sum_{1970-n} \sum_{ALLSLC} \sum_{t} (\Delta C_t \times AREA_{FLCL}) \times 0.02 \times \mathrm{RF\_Base} \times \mathrm{RF\_SN} \times \mathrm{RF\_TX} \times \mathrm{RF\_NSE} \times \frac{44}{28} \times 1000^{-1}$$

N_2O_{FLCL}	=	emissions of N_2O resulting from conversion of forest to cropland from 1970 to year *n* (latest inventory year), kt
ALLSLC	=	all soil polygons where forest land conversion occurs
ΔC_t	=	change in SOC for the t^{th} year after conversion, Mg C ha^{-1} yr^{-1}
$AREA_{FLCL}$	=	area of forest land converted to cropland annually since 1970, ha
0.02	=	conversion of C to N
EF_Base	=	base emission factor, defined as a function of long-term climate normals (growing season precipitation from May to October at the ecodistrict level) (see section A3.4.5)
RF_SN	=	ratio factor for adjusting the effect of source of N on soil N_2O emissions (see section A3.4.5)
RF_TX	=	ratio factor for adjusting soil texture (TX) effect on soil N_2O emissions (see section A3.4.5)
RF_NSE	=	ratio factor for correcting non-growing season soil N_2O emissions (see section A3.4.5)
t	=	time after conversion, year
44/28	=	coefficient converting N_2O-N to N_2O
1000^{-1}	=	converting from Mg to kt

Western Canada

Much of the current agricultural soil in Western Canada was grassland prior to cultivation. Therefore, the conversion of forest to cropland has primarily involved forest that adjoins grassland areas. There is also limited conversion of secondary forest that has grown on former grassland since the suppression of wildfires with agricultural development. Historically, forest conversion has been less prevalent in Western Canada than in Eastern Canada, and fewer comparisons of SOC under forest and agriculture are available in the literature for this region. Ellert and Bettany (1995) reported no difference in SOC values for native aspen forest and for long-term pasture that had remained uncultivated since it was cleared, on an Orthic Gray Luvisol near Star City, Saskatchewan.

CanSIS data provide numerous comparisons of SOC in forest and cropland soils (Table A6.5–9). On average, these data indicate no loss of SOC from forest conversion. This suggests that, in the long term, the balance between C input and SOC mineralization remains similar under agriculture to what it was under forest. It is important to recognize that the northern fringe of western Canadian agricultural areas, where most forest conversion is now occurring, contains marginal land for annual crops, and pasture and forage crops are the primary agricultural uses after clearing. In general, C loss from forest conversion to agriculture is lowest where agricultural land contains forages and pastures.

In Western Canada, no loss of SOC over the long term was assumed from the conversion of forest to pasture and forage crops. Therefore, C losses from land conversion in Western Canada would be from losses of C in above-ground and below-ground tree biomass and coarse woody DOM present in the forest at the time of conversion. Similarly, the average organic N change at sites in Western Canada at least 50 years after conversion from grassland to cropland was +52% (McConkey et al., 2007a), reflecting the substantial added N in agricultural systems compared with forests. However, given the uncertainty over actual soil C–N dynamics after conversion, forest land converted to cropland was assumed not to be a source of N_2O from the soil pool. N_2O emissions are reported wherever biomass burning occurs during conversion (see section A3.5.2.1).

Data Sources

The approach used to estimate the area converted from forest to cropland is described in section A3.5.2.10. The annual area of forest conversion by RU was disaggregated to SLC polygons on the basis of concurrent changes in the area of cropland in SLC polygons. Only polygons that showed an increase in cropland area during the appropriate time period were allocated to forest conversion, and the amount allocated was equivalent to that polygon's proportion of the total cropland increase within the RU.

Uncertainty

The uncertainty associated with C change in each reporting zone was estimated differently for Eastern and Western Canada because of differences in C change estimation methods (McConkey et al., 2007b). For Western Canada, this uncertainty was estimated, despite a mean value of 0 for the C change factor. The assumption was that the uncertainty associated with SOC change after forest land to cropland conversion in Western Canada would follow a similar pattern to that in Eastern Canada.

A3.5.5. Grassland

In Canada, agricultural land reported in the Grassland category is defined as unimproved pasture used for grazing domestic livestock, but only in geographical areas where grassland would not naturally grow into forest if abandoned, i.e. southern Saskatchewan and Alberta and a small area of southern British Columbia. These grasslands developed under millennia of grazing by large animals such as bison, and periodic burning. Essentially, the Grassland category consists of extensively managed native rangeland in Canada.

The primary direct human activities on agricultural grassland in Canada are fire suppression; seeding new plant species in the grassland; and adjusting the amount, duration and timing of grazing by domestic livestock. Methodologies for estimating emissions or removals of CO_2 as a result of direct human activities and for estimating CH_4 and N_2O emissions from natural or prescribed fires on agricultural grassland in Canada are presented in the following section.

A3.5.5.1. Grassland Remaining Grassland

The method for estimating CO_2 is based on the premise that, on long-standing managed grassland, changes in soil C stocks over time occur following changes in soil management that influence the rates of either C additions to or C losses from the soil.

Equation A3.5–21

$$SOC = SOC_{REF} \times F_{MG} \times F_I$$

SOC = soil organic C stock at any given time since management and input change, Mg C ha^{-1}

SOC_{REF} = reference SOC stock, Mg C ha^{-1}

F_{MG} = C stock change factor for management regime, dimensionless

F_I = C stock change factor for input of organic matter, dimensionless

The total area of managed grassland is calculated as follows:

Equation A3.5–22

$$A_n = GLGL_{1990} - \sum_{1990}^{n} GLCL$$

A_n = total area of grassland remaining grassland in the inventory year n, ha

$GLGL_{1990}$ = area of grassland remaining grassland in 1990, ha

$GLCL$ = area of grassland converted to cropland since 1990, ha

Therefore, the net change in SOC because of the management of, and input changes from, grassland remaining grassland can be estimated using the IPCC Tier 1 method as follows:

Equation A3.5–23

$$\Delta C_{GGMineral} = [(SOC_0 - SOC_{0\text{-}T}) \times A]/t$$

$\Delta C_{GGMineral}$	=	net change in SOC due to management of, and input from, grassland remaining grassland, Mg C ha^{-1} yr^{-1}
SOC_0	=	SOC stock in the inventory year, Mg C ha^{-1}
$SOC_{0\text{-}T}$	=	SOC stock *T* years prior to the inventory year, Mg C ha^{-1}
A	=	area of grassland remaining grassland affected by management change or input change, ha
t	=	inventory time period, years (default 20 years)

If no change in management practices or input occurs, C stocks are assumed to be at equilibrium, and the change in C stocks is therefore deemed to be zero.

A number of studies address the effects of grazing versus no grazing on SOC. Although the productivity of heavily grazed pasture is lower, which may lead to a decline in range conditions, this was not found to be related to declines in SOC (Biondini and Manske, 1996). The effects of the grazing regime are complex, because of the effects of grazing on the plant community as well as on the C input to soil from both above-ground and below-ground plant growth (Schuman et al., 2002; Liebig et al., 2005). An additional influence of the grazing regime is the increased return of C in fecal matter as the stocking rate increases (Baron et al., 2002). Bruce et al. (1999) proposed that there was no opportunity to increase SOC from grazing management improvements on extensively managed rangeland in North America.

The addition of organic amendments and inorganic fertilizer increases the productivity of native grasslands (Smoliak, 1965), suggesting that these practices could increase SOC through greater C inputs. However, these practices are basically of academic interest, as the only economically practical management options for semi-arid grasslands are altering the grazing regime, burning, and introducing new plant species (Liebig et al., 2005).

Grasslands managed for grazing in Western Canada in the Brown and Dark Brown soil zones of Alberta, Saskatchewan and British Columbia are occasionally burned by wildfires and by prescribed burning for purposes such as brush management, habitat management, the removal of decadent vegetation, and military training exercises. The burning of managed grassland is a net source of CH_4, CO, NO_x and N_2O.

Equation A3.5–24

$$EMISSION_{BURN} = \sum(AREA_i \times FUELLOAD_i \times C_{F,i} \times G_{EF})\ /1000$$

$EMISSION_{BURN}$	=	emissions of CH_4 or N_2O from prescribed and non-prescribed burning of managed agricultural grassland, kt CH_4 or N_2O
$AREA_i$	=	area of the i^{th} managed agricultural grassland subject to burning, ha
$FUELLOAD_i$	=	average fuel load of the i^{th} managed agricultural grassland subject to burning, Mg DM (dry matter) ha^{-1}
$C_{F,i}$	=	combustion efficiency of the i^{th} managed agricultural grassland subject to burning, fraction, unitless
G_{EF}	=	emission factor of CH_4 (2.7 g CH_4 kg^{-1} dry matter burned) or N_2O (0.07 g N_2O kg^{-1} dry matter burned) (IPCC, 2006)
1000	=	conversion of Mg to kt

Data Sources

As discussed in the section A3.5.4.2, the area reported in the Grassland Remaining Grassland subcategory was estimated using a combination of data from the *Census of Agriculture* and EO, as described in section A3.5.4.1. No detailed and comprehensive activity data are available on management changes on Canadian agricultural grassland over time, except for wild and prescribed fires. Activity data on fire area, fuel load and combustion efficiency for each burning event on managed agricultural grassland were collected through consultations (Bailey and Liang, 2013). Activity data from 2013 to 2015 were updated in 2017 and were assumed to remain constant after the sampling period.

A3.5.6. Wetlands

A3.5.6.1. Peat Extraction

General Approach and Methods

Peat is extracted in Canada for the production of horticultural peat products and related applications, rather than for use as fuel. Since the 1970s, the vacuum harvesting technique has been the dominant method of peat extraction. This technique requires an extensive network of drainage ditches to dry the peat for harvesting by heavy vacuum harvesters. Prior to the implementation of vacuum harvesting, manual block-cutting with shovels was used to extract peat blocks, resulting in a topography of high baulks and low trenches. Although these manual methods are no longer used, numerous abandoned block-cut sites remain in the landscape.

Emissions of CO_2, CH_4 and N_2O from the conversion and management of peatlands for peat extraction were estimated using an IPCC Tier 2 method in accordance based on a combination of the 2006 IPCC Guidelines and the 2013 IPCC Wetlands Supplement (IPCC, 2014). The approach is based on Canadian research and land management practices specific to peat extraction activity in Canada. Emission estimates include on-site CO_2, CH_4 and N_2O emissions, off-site CO_2 emissions from extracted peat, and waterborne C losses of dissolved organic carbon (DOC) from drained and rewetted sites.

Domestic GHG flux studies at peat extraction sites in Canada were reviewed and measurements compiled to develop country-specific emission factors and parameters (Table A6.5–7). Since the majority of flux measurements reported applied only to the growing season, annual CO_2 emission factors were developed by adding measured winter values from Strack and Zuback (2013), which are consistent with the higher winter CO_2 emissions from drained peatlands than from natural peatlands. Annual CH_4 emission factors were developed on the assumption that non-growing season fluxes represent 15% of annual totals derived from natural peatland sites (Saarnio et al., 2007).

Owing to the extraction technology and desired properties of sphagnum peat, preference in site selection is given to open bogs (nutrient poor – ombrotrophic peatlands), which are classified as Other Land under Canada's land categorization framework for the LULUCF sector described in section 6.2 of Chapter 6. Therefore, only approximately 5% of pre-conversion areas meet the definition of the Forest Land category. Emission estimates are separated into the Land Converted to Peat Extraction and Peat Extraction Remaining Peat Extraction subcategories. To calculate the emissions resulting from land conversion, a period of one year is used to represent the land conversion practices of draining and clearing the surface vegetation layer (acrotelm) in preparation for peat extraction. Subsequently, emissions from the ongoing management of peat extraction sites, as well as their decommissioning through abandonment, rehabilitation, or rewetting and restoration, are all reported under the Peat Extraction Remaining Peat Extraction subcategory. The following sections describe the sources of GHG emissions and removals during the various peat extraction phases.

Biomass Clearing and Drainage

At extraction sites, vegetation removal and drainage results in a loss of CO_2 uptake, enhanced peat decomposition, and DOC export, producing increased CO_2 emissions. In drained fields, CH_4 emissions decline substantially, but the drainage ditches, which occupy 5% of the drained area, become CH_4 hot spots (Waddington and Day, 2007). Enhanced peat decomposition also increases N_2O emissions. CO_2 and CH_4 emission factors for drained areas were derived from Canadian studies (Table A6.5–7) but, due to a lack of domestic N_2O measurements, the default emission factor for peat extraction sites in the 2013 IPCC Wetlands Supplement (IPCC, 2014) was used.

Sites that are no longer economical to extract are decommissioned or abandoned. The altered hydrology and peat properties of these sites hinder natural regeneration, resulting in persistent CO_2 emissions (Waddington et al., 2002). However, revegetation occurs more frequently at abandoned block-cut sites than at vacuum-mined sites, although the total vegetation coverage is low and moss regeneration is limited to wetter trench depressions (Poulin et al., 2005). The CO_2 emission factor for abandoned block-cut areas is lower than for areas drained for vacuum harvesting, while the CH_4 emission factor is higher, likely due to greater revegetation and wetter conditions at block-cut sites.

At some abandoned sites, rehabilitation measures are undertaken to establish another type of environment. Given the lack of flux measurements for these sites, the emission factors for drained areas are generally used for rehabilitated areas. However, in sites where trees have been planted, the uptake of CO_2 by trees is calculated on the basis of measurements from a tree plantation study (Garcia Bravo, 2015). Tree plantations may increase CO_2 sequestration in tree biomass, but this does not offset the large CO_2 emissions from drained peat.

Peat Stockpiling and Product Production

Harvested peat is left in stockpiles before being processed into various peat products. Emissions from peat stockpiles are calculated using an exponential decay model for half a year (Cleary et al., 2005). Once it is packaged into products, Canadian peat is transported off-site, largely to the United States, for non-energy uses such as horticulture, where it is assumed to decay in an aerobic environment. Owing to the lack of information on decay rates by end use, it is assumed that all the peat emissions occur in the extraction year. Emissions of CO_2 are calculated based on the estimated total organic C in the peat using a country-specific C fraction parameter (Table A6.5–7) derived from laboratory analyses of pure peat products with a moisture content ranging from 27% to 64% (Hayne et al., 2014).

Rewetting and Restoration

An increasing number of decommissioned sites are rewetted and restored. Rewetting practices increase anaerobic conditions, which reduce peat decay and DOC export, thereby decreasing CO_2 emissions while increasing CH_4 emissions (Strack and Zuback, 2013). Since the 1990s, the moss layer transfer technique has been used in Canada for the restoration of peatlands dominated by *Sphagnum* mosses, with the aim of restoring sites to peat-accumulating ecosystems. This technique consists of rewetting and spreading fields with fresh moss diaspores topped with a layer of straw mulch to support moss regeneration (Rochefort et al., 2003). The long-term monitoring of restoration sites indicates that rewetting and restoration success varies depending on management (e.g. effectiveness of the blocking of the secondary drainage network, timing of restoration procedures and quality of plant material spread) and post-restoration weather conditions (González and Rochefort, 2014). Canadian GHG research at sites restored for 10 years or less has shown high variability among sites, which range from sources to sinks. Given the range of success among sites and the variability in flux measurements, average emission values are used to best represent the net flux at rewetted and restored sites.

Data Sources

An EO mapping approach based on the manual delineation and interpretation of aerial photography, satellite imagery and ancillary data was developed to map the extent of peatland areas disturbed by peat extraction for the circa 1990, 2007, and 2013 time periods. Through image interpretation, the total disturbed area was assigned to one of the following four land management subcategories: active extraction areas, abandoned areas, rehabilitated areas, and restored areas. Geospatial data developed by the Peatland Ecology Research Group and information provided by industry experts were utilized to aid in subcategory allocation. In addition, for a subset of sites, the pre-disturbance land cover class (forest, shrubby or open bog peatland) was determined in order to identify the land category type converted (Forest Land or Other Land).

Annual area estimates were developed using interpolation between mapped time periods and extrapolation after 2013. Annual area estimates for various land management categories were then refined based on secondary data sources. The two main secondary data sources were industry statistics on peatland areas managed for peat extraction in 2015 compiled by the Canadian Sphagnum Peat Moss Association (CSPMA) and a survey of abandoned peat extraction sites in the provinces of Quebec and New Brunswick (Poulin et al., 2005). Secondary data sources were used to provide a basis of comparison check for the total areas converted to peat extraction historically and current production areas, and to complement limitations in the ability of the mapping approach to identify land management subcategories. National peat production statistics were used to represent the annual amount of extracted peat transported off-site (NRCan, 2020).

Uncertainty

Given the increased availability and quality of EO imagery and ancillary information over time, areas for the 2013 mapping time period reduced the uncertainty associated with the overall estimate of the total areas converted for peat extraction. However, considerable uncertainty is associated with identifying land management subcategories. Uncertainty surrounding the 2015 CSPMA industry statistics is linked to the different interpretations of land management category definitions (e.g. restoration) and the incomplete coverage of lands not managed by industry association members.

There is a lack of country-specific GHG measurements for the various categories of decommissioned sites. Therefore, emission factors may not represent the full range and success rates of there habilitation and restoration techniques applied. The large variation in moisture content among peat products may contribute substantially to the uncertainty of estimates of off-site CO_2 emissions from extracted peat.

A3.5.6.2. **Flooded Land**

General Approach and Methods

Following the 2006 IPCC Guidelines, emissions reported under the Land Converted to Flooded Land subcategory (i.e. creation of flooded lands, namely reservoirs) are estimated for all known reservoirs flooded for 10 years or less. Only CO_2 emissions are reported. An IPCC Tier 2 method was used, in which country-specific CO_2 emission factors were developed based on measurements, as described below (Blain et al., 2014). It is believed that the default approach, which assumes that all biomass C would be emitted upon flooding, would overestimate immediate forest conversion emissions from reservoir creation, because the majority of submerged forest biomass does not decay until an extended period of time has passed.

Two complementary estimation methodologies are used to account for GHG fluxes from flooded lands, depending on land conversion practices. When there is evidence of forest clearing and/or burning prior to flooding, immediate and residual emissions from all forest C pools are estimated with CBM-CFS3 (see section A3.5.2.1). Emissions from forest clearing for infrastructure development are reported under the Forest Land Converted to Settlements subcategory. Emissions resulting from the use and disposal of wood products that are harvested before flooding are reported under the Harvested Wood Products category (see section A3.5.3).

In the absence of evidence of forest clearing, it was assumed that all vegetation was simply flooded, leading to CO_2 emissions of a fraction of the submerged C from the surface of the reservoir. The proportion of the area flooded that was previously forested was used to attribute these emissions to either the Forest Land Converted to Flooded Land category or the Other Land Converted to Flooded Land category.

Since 1993, measurements of CO_2 fluxes have been taken above some 57 hydroelectric reservoirs in four provinces: Quebec, Manitoba, British Columbia, and Newfoundland and Labrador (Duchemin, 2006). In most studies, the reservoirs were located in watersheds little affected by human activities, with the notable exception of those in Manitoba. In almost all cases, only diffusive fluxes of CO_2, CH_4 or N_2O (in order of frequency) were measured. Studies on ebullition, degassing emissions and winter emissions are rare and are insufficient to support the development of country-specific emission factors. Measurements of diffusive fluxes above the surface of reservoirs were compiled for the entire country. Among the reservoirs where fluxes were measured, a subset of 34 measurements from 25 reservoirs were selected to develop a national emission curve for the 50-year period following impoundment (Figure A3.5–18). The measurements retained were chosen based on the availability of documentation on measurement procedures and measurement comparability. It is important to note that each of these measurements (data points in Figure A3.5–18) represents, on average, the integration of between 8 and 28 flux samples per reservoir. Non-linear regression analysis was used to parameterize the emission curve.

Equation A3.5–25

$$CO_{2\ rate\ L\_reservoir} = b_0 + b_1 \times In(t)$$

$CO_{2\ rate\ L\_reservoir}$	=	rate of CO_2 emissions from land converted to flooded land (reservoirs), mg m^{-2} per day
b_0, b_1	=	curve parameters, dimensionless
t	=	time since flooding, years

The total CO_2 emissions from the surface of reservoirs were estimated as the sum of all emissions from reservoirs flooded for 10 years or less:

Equation A3.5–26

$$CO_{2\ L\_reservoirs} = \sum(CO_{2\ rate\ L\_reservoir} \times A_{reservoir} \times Days_{ice\ free} \times 10^{-8})$$

$CO_{2\ L\_reservoirs}$	=	emissions from lands converted to flooded land (reservoirs), Gg CO_2 yr^{-1}
$CO_{2\ rate\ L\_reservoir}$	=	rate of CO_2 emissions for each reservoir, mg m^{-2} per day
$A_{reservoir}$	=	reservoir area, ha
$Days_{ice\ free}$	=	number of days without ice, days
10^{-8}	=	conversion factor from mg to Gg

The reservoir area was used as the best available estimate of the area converted to managed flooded lands (reservoirs) although, in reality, reservoirs may contain islands, i.e. emergent land areas. The “ice-free period” was defined as the average number of days between the observed freeze date and the breakup date of ice cover on a body of water (Magnuson et al., 2000). In the case of hydroelectric reservoirs, locations were mapped and estimates of the ice-free period were generated from the Lakes – Ice-Free Period isoline map of Canada (NRCan, 1974).

Emissions were calculated starting with the year of flooding completion. Reservoirs take a minimum of one year to fill following dam completion, unless otherwise confirmed. As CO_2 emissions from the surface of reservoirs are reported only for the 10 years following impoundment, all flooding events since 1980 were used.

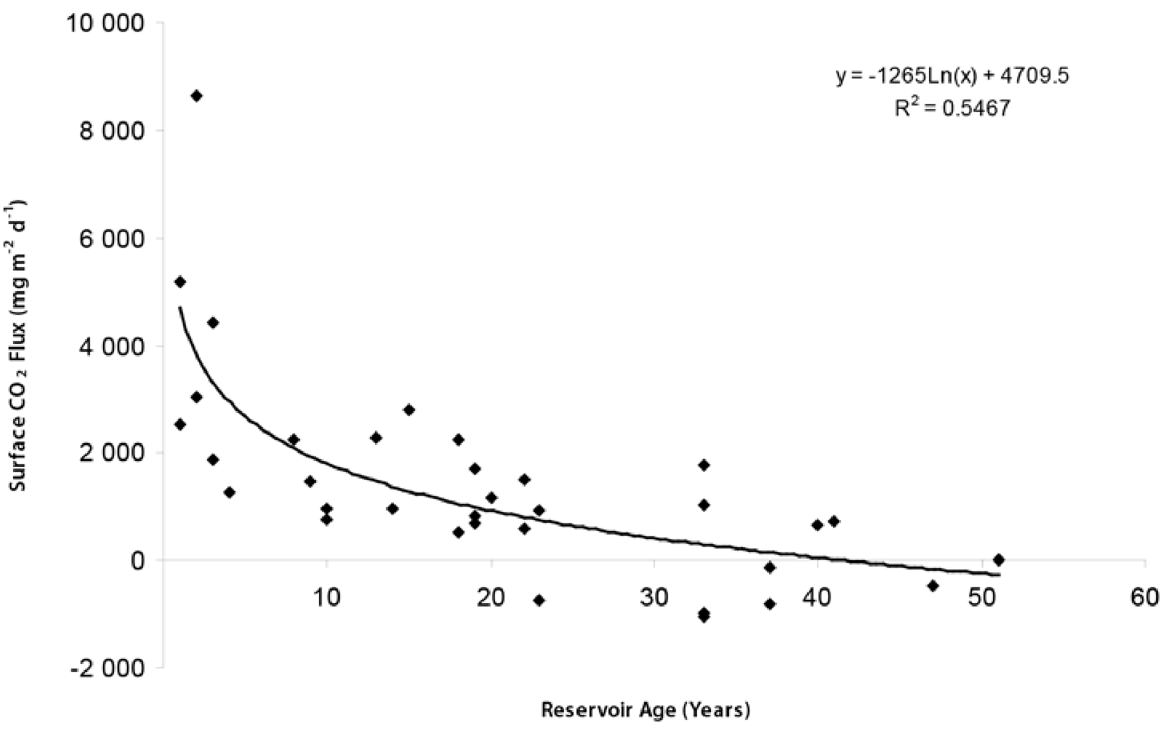
Data Sources

The three main data sources used to develop area estimates were information on forest conversion due to reservoir impoundment in reporting zones 4 (Taiga Shield East) and 5 (Boreal Shield East) (see section A3.5.2.10), the Canadian Reservoir Database (Duchemin, 2002), and official industry numbers derived from industry correspondence[12] (Eichel, 2006).

The Canadian Reservoir Database contains records of 282 hydro reservoirs. Information from provincial and private hydroelectric utilities was accessed to update the database and cross-check the date of reservoir construction and the total reservoir area for all these reservoirs. In some instances, the database reported as new facilities some small, refurbished hydroelectric generation sites in the province of Quebec that entered into production under new ownership. As a result, a separate category was added to the database to document both the original construction and commissioning of a dam and the date when a hydroelectric facility was refurbished without any changes to the reservoir area.

It is important to note that fluctuations in the area of Land Converted to Flooded Land (reservoirs) reported in the CRF tables are not indicative of changes in current conversion rates, but reflect the difference between land areas recently converted (less than 10 years ago) to reservoirs and older reservoirs (more than 10 years), whose areas are thus transferred out of the accounting. The reporting system does not encompass all reservoir areas in Canada, which are monitored separately in the Canadian Reservoir Database.

Figure A3.5–18 **Logarithmic Curve Fit for National Reservoir Emission Factors**



12 Tremblay A, Hydro-Québec. 2010. Personal communication dated November 19, 2010, to Dominique Blain, Environment and Climate Change Canada.

Uncertainty

A temporal curve better reflects the decreasing trends of emission rates after impoundment than a unique emission factor. Hence, the country-specific approach is believed to reduce the uncertainty associated with estimation factors. However, important sources of uncertainty still remain:

- **Seasonal variability** – Some reservoirs display marked seasonal variability in CO_2 fluxes, which is not taken into account in estimate development; anecdotal evidence suggests that algal blooms in the spring could be associated with this variability, especially in reservoirs subjected to anthropogenic nutrient inputs.
- **Reservoir area** – There are variations in reservoir area due to water level fluctuations during the year.
- **Emission pathways** – Potentially important CO_2 emission pathways (e.g. degassing) may be omitted.

A3.5.7. Settlements

This category comprises estimates of removals of CO_2 from Settlements Remaining Settlements (C sinks in urban trees) and emissions from Land Converted to Settlements (conversion of forest land and unmanaged grassland to Settlements). The following sections describe the approaches developed to estimate net C sequestration by urban trees, emissions from the conversion of non-forest land (unmanaged grassland or tundra) to settlements in the Canadian Arctic and Subarctic, and areas of conversion from cropland to settlements. Approaches, methods and data sources for estimating emissions from the conversion of forest land to settlements are covered in sections A3.5.2.1 and A3.5.2.10.

A3.5.7.1. Settlements Remaining Settlements

General Approach and Methods

In Canada, the management and monitoring of urban trees is done at the level of individual municipalities, and there is no centralized authority or organization with responsibility for compiling national-scale urban tree information. Taking into consideration the lack of specific species class information and the considerable resources required to develop such information, an approach based on urban tree crown (UTC) cover area was developed to estimate CO_2 sequestration by urban trees in Canada. The approach involves the sampling of digital aerial photos and high-resolution satellite imagery to estimate the proportion of UTC cover in Canada's major urban areas. The growth of urban trees in Canada was estimated using an IPCC Tier 2A approach (IPCC, 2006):

Equation A3.5–27

$$\Delta C_G = \sum AT \times CRW$$

ΔC_G	=	annual C accumulation attributed to biomass increment of urban trees in settlements remaining settlements, tonnes C yr^{-1}
AT	=	total crown cover area of urban trees, ha
CRW	=	crown cover area-based growth rate (CRW) for urban trees, tonnes C (ha^{-1} crown cover) yr^{-1}

The total urban area of Canada in 2012 was estimated using the boundaries of Statistics Canada's 2011 populated place digital boundary layer,[13] as it was the most nationally consistent delineation of urban areas available. The 1990 urban boundaries were based on Statistics Canada's 1990 polygon layer, which were then manually edited through the visual interpretation of aerial photos and the 1990 GeoCover (MDA-Federal, 2004) ortho-rectified image data set, to reduce known over-bounding errors (Statistics Canada, 2010). The resulting 1990 urban layer represented a smaller total area (1.53 Mha) than the total urban area identified for 2012. Among the 947 population centres (2.42 Mha) in Canada, 69 (1.53 Mha) with populations greater than 30 000 individuals were extracted from the Statistics Canada data set. This subset captured all major Canadian cities and represented 62% and 67% of the total urban area in 1990 and 2012, respectively. Furthermore, this subset includes the urban centres that represented approximately 79% and 76% of Canada's population in 1990 and 2012, respectively (Statistics Canada, 2011; McGovern and Pasher, 2016). While the population centres selected did not completely represent all populated places in Canada, many of the smaller communities that were filtered out are parts of an overall matrix of forest or agricultural land that may be captured under other land categories.

13 Statistics Canada. Populated Place spatial data and information available online at: http://www12.statcan.gc.ca/census-recensement/2011/geo/bound-limit/bound-limit-2011-eng.cfm.

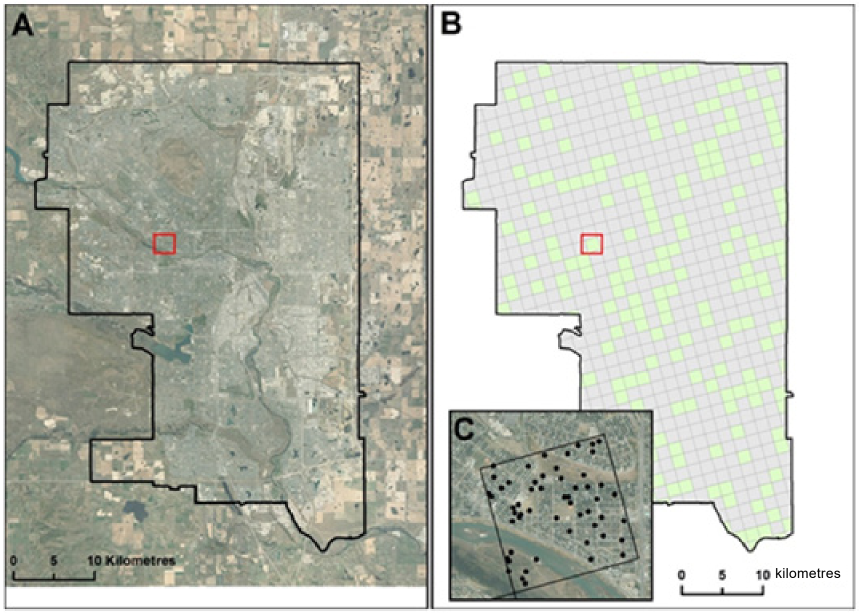
The 69 population centres were spatially allocated to 18 of the 60 RUs (see section A3.5.1). The 18 RUs represented 97% and 99% of the total area and population, respectively, of the 947 population centres. The estimated proportion of UTC cover in each RU was developed using a point-based sampling approach (Pasher et al., 2014). A grid cell approach was used to ensure the adequate spatial distribution of sampling cells (Figure A3.5–19). Random points at a density of 55 points/km^2 on digital aerial photos or high-resolution satellite imagery were interpreted manually and classified into broad categories of tree crown or non-tree crown cover.

The same sampling point locations were used for both the 1990 and 2012 UTC assessments, although sampling cells and points that fell outside the 1990 urban boundaries were not included in order to ensure that sampling was restricted to represent urban areas for that time period. A quality control process was implemented which involved random checks by alternative interpreters or reinterpretation. The percent UTC cover for each RU was calculated as the proportion of all points identified as tree canopy out of the total points that were assessed within the RU. The national-scale UTC estimate was 29% in 1990 and 27% in 2012.

The total crown cover area of urban trees for each RU was estimated by multiplying the % UTC cover by the total urban area estimates for the associated RU in 1990 and 2012. Although urban areas have increased by 6% from 1990 to 2012, the national-scale estimate of crown cover changed little, with some regional variation in trends. Gains in crown cover area (e.g. tree growth and planting) tended to offset losses (e.g. tree removal, mortality and urban land-use change).

The crown cover area-based growth rate (CRW) values for the 18 RUs (see Table A6.5–8) are derived from assessments carried out in 16 Canadian cities using the same methodology used to develop CRW values for the United States. In RUs where cities were not assessed using that approach, values from proxy cities were used based on an ecologically similar Canadian RU, with the exception of RU 41, Pacific Maritime, for which the assessment for the U.S. city of Seattle was used (Steenberg et al., 2021). These assessments take into consideration the tree species, the age of trees, and environmental conditions in each RU to determine gross sequestration rates. Net C sequestration was estimated to be 74% of gross sequestration, accounting for urban tree growth characteristics and tree mortality and decomposition (Nowak et al., 2013). These growth and sequestration rates are applied to the 18 RUs and, as a result, estimates of UTC cover area and the net sequestration rate are the main drivers of overall removal estimates. Interpolation and extrapolation were used to develop a consistent time series for the period from 1990 to the latest inventory year.

Figure A3.5–19 **Sampling Grids and Point Sampling over Georeferenced Aerial Photo**



Note: Background imagery: (A) Calgary, Alberta urban area boundary; (B) 1 km × 1 km grid cells representing a 25% sampling rate with randomly selected grid cells shown in green; and (C) close-up of a single grid cell (20 points km^{-2} sampling). Orthophoto courtesy of City of Calgary.

An analysis on urban trees (Tree Canada, 2019) suggested that 13% of biomass from tree mortality in urban centres is used for firewood. As a result, the volume of firewood harvested from urban trees was estimated by multiplying the C stocks for individual population centres by the estimated 2.4% mortality rate among urban trees taken from Tree Canada (2018) and 13% of C from dead trees was assumed to be used as firewood (Tree Canada, 2019). The firewood was aggregated by RU and assumed to supply a portion of the firewood demand estimated from consumption surveys described in section A3.5.3. To avoid the double counting of emissions in the Settlements and Harvested Wood Products categories, the amount of C transferred to the HWP pool was attributed to the difference between the gross and net annual sequestration rates estimated by the i-tree model. As a consequence, the C sink under Settlements is apparently increased due to the combustion of this C as residential firewood being reported under Harvested Wood Products.

Uncertainty

The uncertainties associated with the estimates of urban area, UTC cover and the C sequestration rate all contribute to the overall uncertainty associated with estimated CO_2 removals by urban trees. The result of these combined uncertainties using a Tier 2 Monte Carlo analysis associated with each RU using the field data on urban trees collected in Canada and the city of Seattle, provides an estimated total uncertainty of 38% for 1990 and 2012. This uncertainty estimate does not include the estimation error related to the use of biomass equations or conversion factors or to measurement error (Nowak et al., 2013).

The uncertainties associated with the 1990 and 2012 urban areas were not quantified by Statistics Canada. An error estimate of 10% was used for the 2012 urban area following the approach used in the United States' 2012 national GHG inventory report (U.S. EPA, 2013). The error associated with the 1990 urban area estimate was assumed to be slightly higher at 15% than that for 2012, based on expert judgment. This approach is similar to the uncertainty estimate for the boundary delineation (15%) used for developing forest conversion estimates (Leckie, 2011).

The uncertainty associated with UTC estimates was based on the standard error in the sampling approach calculated for each sampling period (1990 and 2012). Standard errors for the UTC estimates were low (0.2% for the national UTC estimate) given the very high number of sampling points used.

A3.5.7.2. Cropland Converted to Settlements

Data Sources

Urban and industrial expansion has been one of the main drivers of the conversion of cropland to settlements in Canada. The areas of cropland converted to settlements were estimated based on the land-use maps for 1990, 2000 and 2010 developed in Huffman et al. (2015a). Areas converted during the 1990–2000 and 2000–2010 periods were calculated by spatial analysis for each RU, divided by the number of years, in order to develop constant annual conversion rates. Converted areas were extrapolated after 2010. The total area of cropland converted to settlements during the 1990–2000 and 2000–2010 time periods was 184 kha and 115 kha, respectively, with the majority of change due to urban expansion in reporting zones 7 (Mixedwood Plains) and 11 (Sub-humid Prairies). This is largely due to urban expansion in the main populated centres, such as Toronto, Hamilton, Oshawa, Montreal, and Edmonton.

Uncertainty

Given that the highest conversion rates are caused by urban expansion, an independent assessment of converted areas was conducted by comparing the land cover in each map against the visual interpretation of ortho-rectified Landsat imagery over urban centres. The sampling strategy used in this assessment was to perform the analysis on five main census metropolitan areas (CMA[14]), which account for 45% of the total area change from cropland to settlements. Polygons from the 2011 census were used to define the boundary of each CMA, and over 400 stratified random points were used to verify the land cover class in areas in which there were examples of either change or no-change from cropland to settlements, separated by a minimum distance of 1 km, to avoid statistical bias. The minimum mapping unit for the accuracy analysis was defined as a circle with a radius of 100 m to prevent errors due to the presence of noise in each classified map. The class in each location was assigned based on the class of the majority of the pixels, to account for changes in land use. An overall accuracy of 80% and 84% was obtained for the areas of change computed from these maps, which concurs with the accuracy assessment carried out by Huffman et al. (2015a).

14 This term has been defined by Statistics Canada as the area consisting of one or more neighbouring municipalities with a population of 100 000 inhabitants or more.

A3.5.7.3. Grassland Converted to Settlements

General Approach and Methods

Nearly half of Canada's land mass is in the Arctic and Subarctic regions and includes all land categories (IPCC, 2006), excluding Cropland. An approach was developed specifically for capturing the emissions associated with land-use changes by identifying the drivers and the amount of biomass C stocks in this vast and remote landscape. It included the following components: (i) manual digitizing of land-use polygons in Canada's Arctic/Subarctic for 1990, 2000 and 2010 based on ortho-rectified Landsat imagery and the assessment of land-use changes over about 359 million hectares, including areas in reporting zones 1 (Arctic Cordillera), 2 (Northern Arctic), 3 (Southern Arctic), 4 (Taiga Shield East), 5 (Boreal Shield East), 8 (Hudson Plains), 10 (Boreal Plains), 13 (Taiga Plains), 16 (Boreal Cordillera), 17 (Taiga Cordillera) and 18 (Taiga Shield West), north of 60°N latitude; and (ii) the estimation of above-ground biomass based on field samples taken in Canada's Arctic/Subarctic regions between 2004 and 2010, covering the northern part of the Boreal Cordillera, Taiga Plains, Taiga Shield East, Taiga Shield West, Southern Arctic, Northern Arctic, and Arctic Cordillera.

A comprehensive, wall-to-wall analysis of land-use circa 1990, 2000 and 2010 was carried out based on image interpretation followed by manual digitization of the sites undergoing change (McGovern et al., 2016). A wide range of human disturbances such as airstrips, roads, power lines, seismic lines, urban areas, mines, reservoirs and even smaller features like well sites and some roadside clearings were identified using snow- and ice-free imagery. The analysis of existing GIS data sets denoting the occurrence of anthropogenic development were used to guide the search for areas with a high probability of land-use change. Mapping was then expanded outwards from these regions based on the observation of additional disturbances. The resulting spatial data set provided the most comprehensive and complete mapping product for human disturbances in Canada's northern regions, and builds on previous boreal disturbance mapping activities conducted by ECCC. An interpretation guide similar to that produced for the Canadian Forest Service (Dyk et al., 2015) was used to guarantee consistency in the detection, digitization and categorization of disturbances. A total of 1135 scenes were used for the interpretation process (395 for 1990, 348 for 2000 and 392 for 2010).

Land-use change was derived from the difference in polygon areas for each date, providing the area of changes between time periods (i.e.1990–2000, 2000–2010), which was then divided by the total number of years in the time period to produce a constant annual rate of change. The same annual rate of land-use change was applied tor the years prior to 1990 and after 2010. The pre-conversion land-use type for each of the land-use change polygons was based on available land cover maps (Wulder et al., 2008; Hermosilla et al., 2016), visual interpretation and vegetation indices for concurrent imagery to avoid including areas in land-use categories other than Settlements. Furthermore, deforestation events above 60 degrees latitude were also used to confirm that areas determined to be forest converted to settlements were excluded, to avoid double counting.

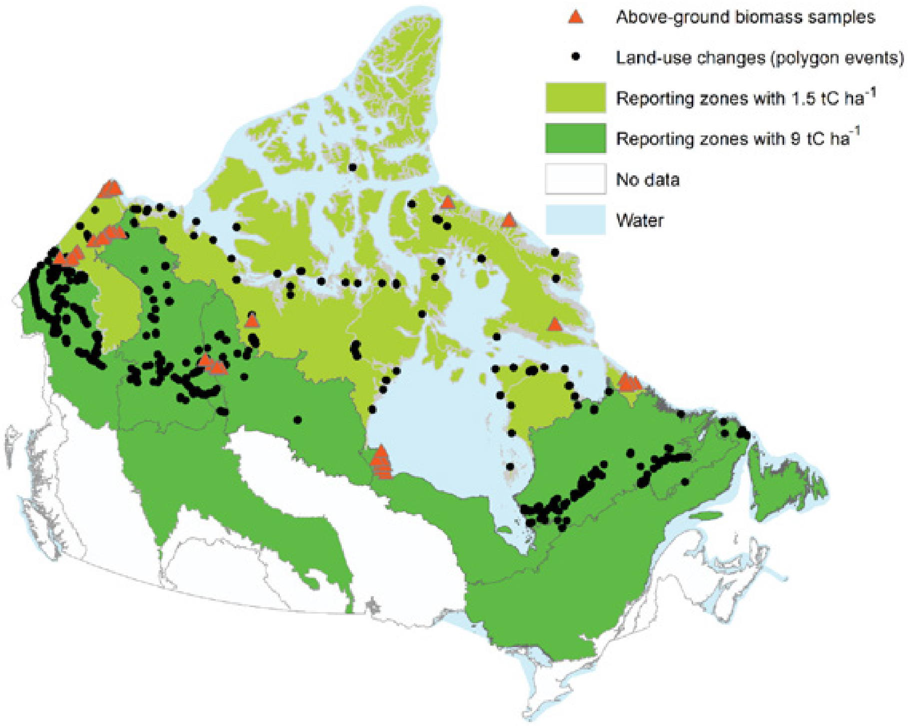
Biomass losses were derived from the statistical analysis of field samples obtained in surveys between 2004 and 2011 of the Canadian North (Figure A3.5–20). Over 116 samples were collected from different land-cover types (e.g. shrubs, grass tundra, wetland, forest and barren land) in eight reporting zones. The vegetation in this region consists of forest patches in the Boreal Cordillera and Taiga Plains, but predominantly low vegetation composed of sparse shrubs, mixed grass-dwarf shrub, lichen, moss tussock sedge, bare soil and Arctic willow tundra in the remaining reporting zones. Due to the diversity of vegetation types and landscapes across this region, field samples from forest were excluded and the remaining samples were grouped into two classes: high and low vegetation. This grouping was based on the fact that, after statistical examination of above-ground biomass values, significant variability was found in the sampled vegetation types between reporting zones. As an initial implementation, the mean of the samples in reporting zones 1 (Arctic Cordillera), 2 (Northern Arctic), 3 (Southern Arctic) and 17 (Taiga Cordillera) was used to obtain a single value of above-ground biomass (1.5t C ha^{-1}) that was applied to all these reporting zones—i.e. areas with "low" vegetation types. Similarly, a single average value (9 tC ha^{-1}) from all samples in the remaining reporting zones (Taiga Plains, Taiga Shield West, Boreal Cordillera and Hudson Plains) was used and applied to the remaining areas—i.e. areas with "high" vegetation. Reporting zones with land-use change data but without field samples (i.e. Taiga Shield East, Boreal Shield East and Boreal Plains) were assigned to either of the two groups of low or high vegetation based on an analysis of vegetation indices. Emissions from land-use changes were estimated by multiplying the annual area affected by land-use changes by their respective biomass loss factor to obtain C stock changes. Annual area rates and emissions for the years after 2010 were extrapolated from the 2000–2010 period, assuming a constant yearly rate.

The biomass factor obtained for each of the two vegetation groups was assessed based on the vegetation characteristics of each ecozone (Marshall et al., 1999) and values in the literature (Shaver and Chapin, 1991; Hudson and Henry, 2009; Gould et al., 2003) and was also compared with the values reported in the 2006 IPCC guidelines for the boreal and cool temperate regions (IPCC, 2006). All land-use change activities involved the conversion of Arctic tundra vegetation to settlements, and all pre-conversion biomass C was deemed emitted upon clearing.

Uncertainty

The error propagation approach was used to estimate uncertainty using a 95% confidence interval. The percentage of uncertainty for the above-ground biomass volume was 70% for ecozones with low vegetation and 80% for all the other ecozones, based on the coefficient of variation. The uncertainty associated with the total area affected by land-use change was estimated to be 30%, based on random sampling and image interpretation. A 20% uncertainty was used for the C content, estimated to be 50% of the dry biomass weight, based on the IPCC guidelines (IPCC, 2006). Using these values, an overall uncertainty of 87% was estimated for this category.

Figure A3.5–20 **Location of Land-Use Change Events and Field Samples of Above-Ground Biomass in Canada's North**



Note: More southerly reporting zones are attributed to the 9 tC ha^{-1} biomass class, as some sites border on the northernmost boundary of these reporting zones.

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