Measuring Externalities in Canadian Agriculture: Understanding the Impact of Agricultural Production on the Environment

Report prepared for CAPI

by

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Executive Summary

Canada is known to be a country with an abundance of land, water and natural resources making it one of the leading producers and exporters of both renewable and non-renewable resources. Canadians are blessed with a comfortable and growing standard of living that is the envy of the world. With a thriving agriculture and agri-food sector, Canada continues to produce and export high quality food and agricultural products to countries around the world. Canada's farmers are leading edge, having adopted innovative technologies and management practices that contribute to ever growing yields and new crop varieties. Canada has also become one of the most efficient livestock producers in the world. With the help of world class research and education, Canada's farmers are well positioned to feed the world's growing population with increasingly healthy and sustainable food products.

However, Canada's "natural capital" is not infinite.[1] While we are able to produce a growing quantity of agricultural and food products year after year, there are concerns over whether we are doing so in a sustainable fashion. Agricultural production that is not sustainable will have impacts on the environment far into the future, such as from air and water pollution, soil erosion and loss of biodiversity and wildlife habitat. It can also impact human health. The sector is also cited as a source of Greenhouse Gas (GHG) emissions, contributing to climate change. Future generations will be impacted if the growth in agricultural production today comes at the cost of future environmental degradation and natural capital loss in Canada.

Understanding the extent to which agricultural production is impacting the environment (and human health) requires metrics that measure these impacts. For the purpose of this study, these impacts are defined as externalities. [2] They can be both positive or negative and are often described as "external costs" or "benefits".

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[1] Natural capital is defined as the world's stock of natural resources, which includes geology, soils, air, water and all living organisms. Some natural capital assets provide people with free goods and services, often called ecosystem services.

[2] The first discussion around 'externalities' was published in a paper by R. Coase, "The Problem of Social Cost" in the Journal of Law and Economics, October 1960.

According to the OECD:

> Environmental externalities refer to the economic concept of uncompensated environmental effects of production and consumption that affect consumer utility and enterprise cost outside the market mechanism.[3]

In other words, externalities are costs or benefits that are imposed or accrue to people who are not directly involved in the transactions that generate these costs or benefits, i.e. society at large. They are produced "outside the market-place" and hence their value is unknown and not factored into everyday production decisions by farmers. If it is important to society to address these externalities, then there must be mechanisms for incenting farmers to do so, through subsidies, regulations, education and moral suasion. The objective of this report is to place an economic value on agricultural externalities in Canada so that we can have a better sense of the magnitude of the impact agriculture has on the environment, human health and Canada's natural capital and resources. This will then allow us to determine the potential policy measures needed for addressing them.

Methodology

Measuring externalities in a country as large and diverse as Canada is a challenging task. In this report, we have followed the three main steps in economic valuation, namely to: i) identify the externality, ii) quantify, and iii) monetize or valuate.

We rely mainly on data from Agriculture and Agri-food Canada's (AAFC) Agri-Environmental Indicators Report (2016) for quantities of externalities from agriculture.[4] Economic valuation of these quantities relies on using existing studies in the literature, both Canadian and international, and transferring the estimates to a new context through value transfer methods. Many of these studies

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[3] OECD, https://stats.oecd.org/glossary/detail.asp?ID=824.
[4] AAFC, "Agri-Environmental Indicators Report # 4, 2016.

are from international examples, but international comparisons are not always appropriate.

Limitations

While we have attempted to be as comprehensive and rigorous as possible, there are limitations with each of these steps that should be kept in mind when reading the report and interpreting the results. First, there are numerous externalities for which we could not find appropriate biophysical data to quantify the impact from agriculture. For example, we were unable to determine the net impacts of agriculture on wetlands conversion. Agricultural land is contiguous to and includes large areas of wetlands which provide positive benefits to society such as wildlife biodiversity, water filtration and storage etc.[5] However, agricultural land conversion is a major driver of wetland loss. Consequently, the estimates of externalities in this report are unfortunately only a partial accounting of the external costs of agriculture on the environment and human health.

Second, there are significant data gaps and uncertainties on the biophysical quantities associated with agriculture's impact on the environment and human health. Determining the external impacts of agriculture requires a counterfactual scenario for comparison. While we have attempted to be as consistent as possible across the different externalities, there are some differences in the implied scenarios for each externality as described in each section. The data this report draws heavily from, AAFC's Agri-Environmental Indicators report (2016), while essential for estimating externalities in agriculture, still leave some gaps in information, and these are more fully described in the report.

Third, there are limitations with the methods used to value these externalities. The main difficulty in estimating their economic value is the lack of a market price. Agricultural externalities such as fertilizer runoff or air pollution are not traded in

[5] For a description of the benefits of wetlands to society and the environment, see CAPI's recently published report on the Contribution of Wetlands to Sustainable Agriculture, available here: https://capi-icpa.ca/wp-content/uploads/2019/11/2019-10-09-CAPI-Wetlands-CAPI-Doctoral-Fellows-2017-19-group-paper_WEB.pdf.

a market, and hence we cannot directly observe their marginal cost to society. To place an economic value on externalities, this report relies on the value-transfer method which applies existing value information to the appraisal of each biophysical, economic, temporal and/or spatial situation. Value-transfer methods were chosen due to time and budget constraints associated with conducting primary valuation research. However, these approaches are best viewed as providing an order of magnitude estimate. Another limitation of the economic value step is that we use per unit values that are constant over time after adjusting for inflation. Thus, we do not account for changing resource scarcity or socio-demographic characteristics such as income that could influence the per unit value of externalities over time. Finally, we have also used other valuation approaches, such as the replacement cost method for nitrogen (N), that do not capture the full value of the externality, but rather reflect the costs of removing N from the environment.

Considering these limitations, we recommend that the numbers described in this report be viewed as preliminary estimates of the externalities or external costs or benefits from Canadian agriculture rather than the definitive account.

Results

The results are summarized in Table 0.1 below. Estimates are for the three Prairie provinces in Western Canada and Central Canada only. Under the external cost to air category, we considered GHG emissions, ammonia (NH_3) emissions and particulate matter (PM) emissions. Valuation for GHG emissions from agriculture ranged from $1.7 billion in 1981 to $1.5 billion in 2011, a 10% decline over the period. Values for NH_3 emissions from agriculture were available only for the years 1981, 2006, and 2011. The external cost of NH_3 from agriculture rose from $1.3 billion in 1981 to $1.7 billion in 2006 before falling again to $1.5 billion in 2011. There was an estimated 14% increase in these costs between 1981 and 2011. Costs from PM emissions plummeted significantly between 1981 and 2011. Total costs in both Central and Western Canada were estimated at $4 billion in 1981, dropping to $1.6 billion in 2011, a decline of about 60%. Western Canadian agriculture contributed the most to this decline in PM emissions over time as a result of more sustainable tilling practices.

Generally, the external cost estimates for water pollution parameters demonstrate a deteriorating trend between 1981 and 2011. This cost increased in all provinces over time. The cost of N water pollution rose by about 40% between 1981 and 2011. Specifically, this cost rose from $706 million in 1981 to $985 million in 2011. The external cost of Phosphorous (P) water pollution rose from $48 million in 1981 to $55 million in 2011, a 14% increase. External costs from pesticide water pollution deteriorated the most by 61% increase over the period from $539 million in 1981 to $869 million in 2011. Only the cost of coliform water pollution declined by 3% between 1981 and 2011, from $43 million to $42 million in 2011.

We estimate the cost of soil erosion for different erosion risk level classifications as defined by AAFC (2016). The cost from soil erosion fell by 28% from $2.84 billion in 1981 to $2.05 billion in 2011. Western Canada experienced a 32% reduction in the costs from soil erosion while Central Canada experienced a 12.7% decline over the period (Table 4.8). Each province included in these estimates experienced declining external costs from soil erosion except for Quebec, which saw a 0.6% increase.

The estimated cost of biodiversity and wildlife habitat damage increased about 1.3% between 1981 and 2011 in Western Canada, while Central Canada showed an improvement over time with a 14% decline in cost over the same time period (Table 4.9). The total cost for biodiversity and wildlife habitat damage for both regions in 1981 was $286 million and $253 million in 2011, reflecting a decline of about 12% (Table 0.1).

We considered the control of soil erosion, the provision of wildlife habitat, landscape aesthetics and nutrient recycling as positive externalities from agriculture. This list of positive indicators is not exhaustive, but includes those indicators for which data were available and could be estimated with relative ease. Carbon sequestration, which is a well-known positive externality, is not included in this report because the estimates for GHG emissions we used were net of the carbon that has been stored in soils over this period. Further analysis is needed to estimate the external benefits from this source.

The positive externalities or benefits from erosion control were estimated at $2.05 billion in 1981 and $2.02 in 2011, a 1.4% reduction. Most of these benefits took place in Western Canada, where estimates rose from $1.7 billion in 1981 to $1.8 billion in 2011. This compares with Central Canada, where benefits from erosion control were estimated to fall from $365 million in 1981 to $256 million in 2011. In contrast, Central Canada's contribution to benefits from biodiversity and wildlife outweighed that of Western Canada. The estimate for positive externalities related to wildlife habitat and biodiversity in both Central and Western Canada rose from about $38 million in 1981 to $32 million in 2011. Western Canada contributed $3.8 million in 1981 and $3.6 million in 2011 while Central Canada contributed $33.9 million and $28.8 million, respectively (Table 5.1).

Landscape aesthetics are considered a positive externality as a benefit to society and provide large benefits. The total benefit for both regions was $4.6 billion in 1981 and $4.5 billion in 2011 (Table 0.1). About 85% of this estimate originated from Western Canadian agriculture.

Summary and Implications

Table 0.1 below, which presents a summary of the findings from this report, can also be viewed as a set of policy priorities for addressing agricultural externalities in Canada. However, we are missing an important component—the cost of mitigation. In order for policies that address these externalities to be efficient, they must reduce external costs at the lowest possible unit cost of damage avoided. While soil erosion appears to be the most serious externality in Canada, this does not necessarily imply that it should have the highest policy priority. What is an important consideration is the marginal abatement cost for each of these externalities, i.e., the cost to society of reducing one unit of the externality. With this information, we could rank each externality by the ratio of marginal damage to society to marginal cost of abatement, which would provide us with an efficient place to start in terms of addressing the externalities from agriculture. In this report, we have calculated the numerator of this all-important ratio (i.e the cost to society). But in future research, we need to address the denominator (i.e. the cost of mitigation).

Another interesting policy implication from these results is the extent to which these externalities vary across regions and provinces. Apart from water quality damages, which are an issue in each province, soil erosion is the largest source of external costs in the Western provinces while GHG emissions are the largest source in Central Canada. This heterogeneity suggests that provincial-level policy approaches will most likely differ from federal-level policy approaches to address the issues most relevant for each region. Further, with the exception of GHG emissions, each of the externalities we address in this report have localized effects, which suggests provincial-specific targeting will be important. The fact that GHG emissions are damaging regardless of their origin makes it the most difficult externality to solve—Western provinces will have little appetite to reduce GHG emissions when all their work could be undone by inaction in Central Canada (or vice versa). This suggests that a national approach to GHG emissions may be the most efficient solution.

TABLE 0.1. Negative, positive, and net environmental externalities over time for the Western and Central Provinces (millions of 2012 dollars)

	Negative externalities							
Externality	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**Percentage change (1981-2011)**
GHG[6]	1,679	1,628	1,609	1,768	1,659	1,628	1,503	-10%
Ammonia (NH_3)	1,319					1,696	1,499	14%
PM	3,989	3,651	3,278	2,986	2,544	2,061	1,601	-60%
N-water	706	857	806	810	942	981	985	39%
P-water	48	52	52	54	57	56	55	14%
Pest-water	539	592	655	701	754	813	869	61%
Coliform-water	43	41	42	42	42	44	42	-3%
Soil erosion	2,843	2,950	2,828	2,733	2,637	2,226	2,049	-28%
Wildlife/biodiversity	286	274	266	271	266	264	253	-12%
Total negative	**11,452**	**10,043**	**9,535**	**9,365**	**8,901**	**9,768**	**8,856**	**-23%**
	Positive externalities							
Externality	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**Percentage change (1981-2011)**
Wildlife habitat	38	36	35	35	35	34	32	-6%
Landscape aesthetics	4,607	4,739	4,739	4,748	4,705	4,693	4,506	-4%
Total positive	**4,644**	**4,774**	**4,773**	**4,783**	**4,739**	**4,728**	**4,539**	**-2%**
Total net	**-6,808**	**-5,269**	**-4,762**	**-4,582**	**-4,162**	**-5,040**	**-4,318**	**-37%**

[6] The values shown in Table 0.1 are the cross-Canadian averages with different parts of the country having vastly different externality values due to farming methods, soil types, and other factors like farming intensity and climate.

1 Introduction

Canada is known to be a country with an abundance of land, water and natural resources making one of the leading producers and exporters of both renewable and non-renewable resources. Canadians are blessed with a comfortable and growing standard of living that is the envy of the world. With a thriving agriculture and agri-food sector, Canada continues to produce and export high quality food and agricultural products to countries around the world. Canada's farmers are leading edge, having adopted innovative technologies and management practices that contribute to ever growing yields and new varieties. With the help of world class research and education, Canada's farmers are well positioned to feed the world's growing population of increasingly wealthy consumers into the future.

However, Canada's natural capital is not infinite.[7] While we are able to produce a growing quantity of agricultural and food products year after year, there are concerns over whether we are doing so in a sustainable fashion. Agricultural production that is not sustainable will have impacts on the environment, such as from air and water pollution, soil erosion and loss of biodiversity and wildlife habitat. It is also cited as a source of GHG emissions contributing to climate change. It can also impact human health. Future generations will be impacted if current production growth comes at the cost of future environmental degradation and natural capital loss in Canada.

To understand the extent to which agricultural production is impacting the environment requires metrics that measure these impacts. For the purpose of this study, these impacts are defined as "externalities".[8] They can be both positive or negative. In other words, "externalities" are costs or benefits that are imposed or accrue to people who are not directly involved in the transactions that generate these costs or benefits, i.e. society at large. They are produced "outside the

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[8] The first discussion around 'externalities' was published in a paper by R. Coase, "The Problem of Social Cost" in the Journal of Law and Economics, October 1960.

market-place" and hence their value is unknown and not factored into everyday production decisions by farmers. If it is important to society to address these externalities, then there must be mechanisms for incenting farmers to do so, through subsidies, regulations, education or moral suasion. The objective of this report is to place an economic value on agricultural externalities in Canada, which will help increase our understanding of the magnitude of the impacts agricultural production has on the environment, human health and Canada's natural capital and resources. This will then allow us to determine the potential policy measures needed for addressing them.

2 What are Externalities?

For proper interpretation of the analysis to come, it is important to be clear about what we mean by externalities. According to the OECD:

> Environmental externalities refer to the economic concept of uncompensated environmental effects of production and consumption that affect consumer utility and enterprise cost outside the market mechanism.[9]

In restricting ourselves to externalities "outside the market mechanism," we are focusing only on non-pecuniary externalities. In other words, we neglect situations where actions of one party influence the prices faced by another, such as when an importing country refuses to accept Canadian agricultural products, which are referred to as pecuniary externalities.

In other words, externalities are costs or benefits that are imposed or accrue to people who are not directly involved in the transactions that generate these costs or benefits, i.e. society at large. They are produced "outside the market-place" and hence their value is unknown and not factored into everyday production decisions by farmers.

In the agricultural sector, the most significant externalities impact air quality, soil quality, water quality, wildlife, and greenhouse gas (GHG) emissions—all of

[9] OECD, https://stats.oecd.org/glossary/detail.asp?ID=824.

which have direct or indirect impacts on human and/or animal health. In crop production, many of these externalities are driven by the impact of input use, such as fertilizer. Increasing fertilizer application rates beyond the uptake threshold results in leaching, both into nearby water sources and into the air. For this reason, externality valuations are closely related to fertilizer emission intensity. Figure 2.1 and Figure 2.2 show variations in fertilizer emission intensity in Western, Central and Eastern Canada in 2011.[10] Because of soil type variation, differences in crop selection, and differences in climate, fertilizer emissions intensity is much greater in Eastern Canada than Western Canada.

[10] In this example, Western Canada includes British Columbia, Alberta, Saskatchewan and Manitoba; Central Canada includes Ontario and Quebec, and Eastern Canada includes New Brunswick, Nova Scotia, Prince Edward Island and Newfoundland.

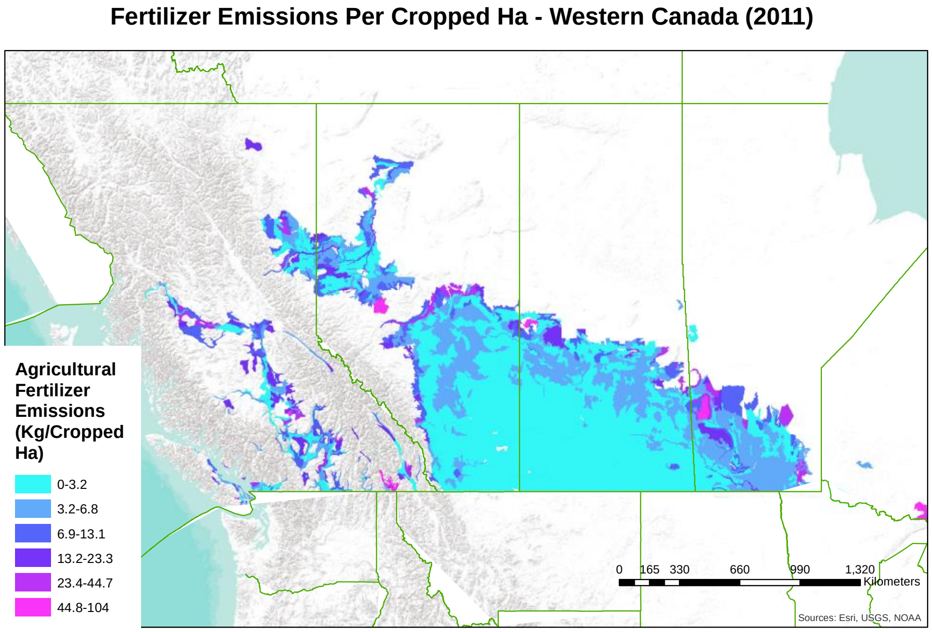

Figure 2.1. Fertilizer emission intensity in Western Canada, 2011

Source: AAFC and Statistics Canada

Fertilizer Emissions Per Cropped Ha - Central and Eastern Canada (2011)

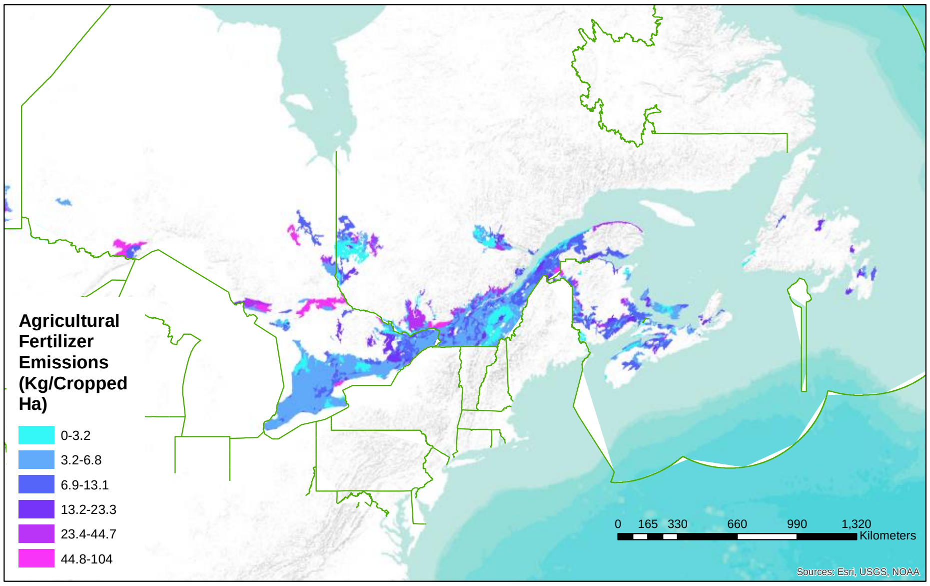

Figure 2.2. Fertilizer emission intensity in Central and Eastern Canada, 2011

Source: AAFC and Statistics Canada

Table 2.1 aggregates the spatial data in the previous figures by province, offering a different perspective on the variation in fertilizer emissions intensity across provinces. Saskatchewan ranks lowest, with an average of 2.9 kg/cropped hectare, followed by Alberta and Manitoba. Both Quebec and Ontario exhibit much higher fertilizer emissions intensity by comparison.

TABLE 2.1. Fertilizer emissions intensity by province, 2011

Province	Kg/ha	Kg/Cropped ha	Mean percent agricultural land
AB	1.7	4.1	64
SK	1.8	2.9	78
MB	2.3	5.7	60
ON	2.7	6.5	54
QC	2.7	6.5	54
Canada	1.8	5.4	54

Source: AAFC and Statistics Canada

With the exception of GHG emissions, the external costs of agricultural production depend critically on the proximity of agricultural activity to population centers. Air pollution, especially particulate matter (PM) and ammonia (NH_3), are significantly more damaging to human health when generated near population centers (Muller and Mendelsohn, 2007). A similar relationship holds for water pollution, although pollutant transport via rivers and streams has the potential to impact a much larger area. As urban centers expand, especially those in the midst of highly concentrated agricultural areas, the damage created by these types of externalities will continue to increase.[11] Figure 2.3 and Figure 2.4 show the areas of agricultural concentration in Western, Central and Eastern Canada, which in many cases are adjacent to fast-growing population centers.

[11] This is reminiscent of an externality example popularized by Coase (1960), i.e., if someone moves in next to a factory that emits smoke as a by-product of production, who should be responsible for paying to reduce the smoke to the socially optimal level—the factory emitting the smoke or the person who decided to move next to the factory?

Percent of Western Canadian Land Being Used for Agriculture in 2011

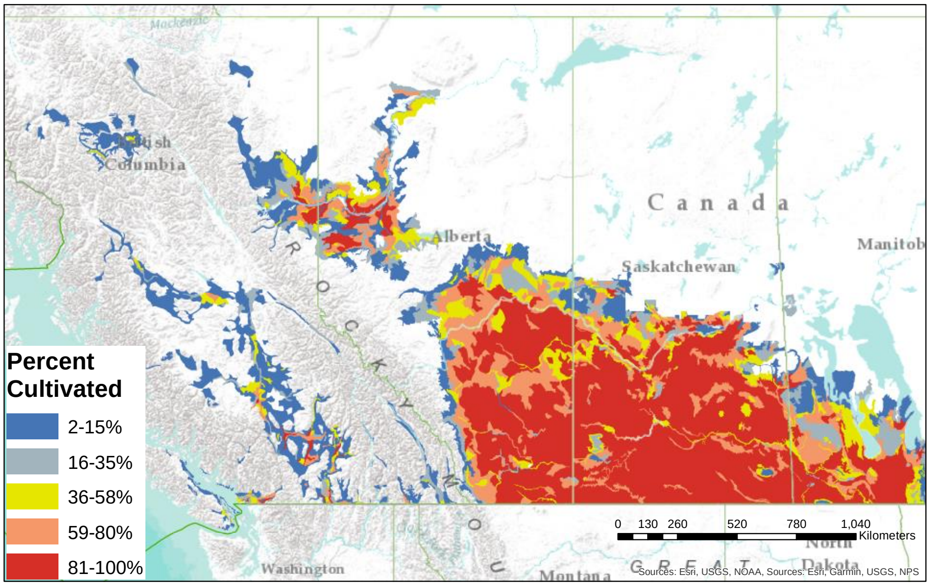

Figure 2.3. Agricultural land use intensity in Western Canada, 2011

Source: AAFC and Statistics Canada

Percent of Central and Eastern Canadian Land Being Used for Agriculture in 2011

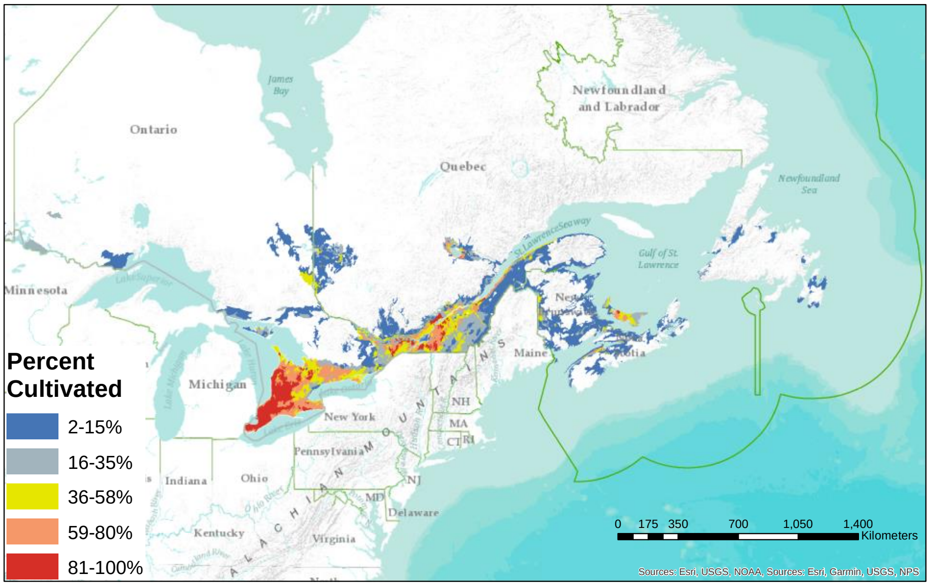

Figure 2.4. Agricultural land use intensity in Eastern Canada, 2011

Source: AAFC and Statistics Canada

Externalities are most often thought of as negative influences, but in the agricultural sector, there are several examples of positive externalities. In this report, we highlight the role of wildlife habitat and biodiversity improvements, the value of landscape aesthetics, and the strength of rural communities as positive externalities.

While the sections that follow are focused primarily on identification, quantification, and valuation of environmental externalities relating to agriculture, we do wish to make a few observations regarding policy. First and foremost, we would like to stress that the goal of policy designed to mitigate externalities is not to eliminate the externality entirely, but to reduce externalities to the *socially optimal level*, which incentivizes production to occur at the point where marginal social benefit is equal to the marginal social cost. The damage from over applying N fertilizer is significant, but that does not imply that the socially optimal response is to ban fertilizer application outright. Instead, the yield increases resulting from fertilizer application must be appropriately balanced against the associated environmental damage or externalities from its use.

The regulation of agricultural externalities also poses unique policy challenges. The connection between farm management practices and externalities is opaque even in the most textbook of circumstances. This is especially the case for GHG emissions, where emissions can vary substantially depending on weather, soil characteristics, input absorption, and crop selection. Even if this relationship were purely deterministic, the difficult problem of observation remains: how can we regulate what we cannot accurately measure (or attribute)?

In developed economies, the most popular way to circumvent the observation problem and mitigate agricultural externalities through incentivization and subsidization: offering payments (or cost-sharing) for producers to adopt farm management practices associated with reductions in harmful pollutants or increases in positive externalities. But to design efficient mitigation programs and policies, we must first have a sound understanding of the monetary damages and benefits induced by agricultural externalities. In the following sections, we combine publicly available data provided by Statistics Canada and Agriculture and Agri-Food Canada (AAFC) with environmental valuation studies to provide a comprehensive valuation of these externalities.

3 Methodology

To measure environmental externalities in Canadian agriculture, we followed three steps. First, we identified the set of externalities for which we had data of sufficient quality. Second, we used secondary data to measure the physical quantities of each externality at the annual-provincial level, e.g. tonnes of GHGs produced by the agricultural sector in Saskatchewan in 2016. Third, we used the benefit-transfer method to estimate the monetary value of the externality measured in the second step. For the measurement and valuation steps, each externality requires subtle variations, which we outline in “measurement” and “valuation” subsections for each externality.

In the identification step, we relied on a recently produced report from AAFC describing trends in key agri-environmental indicators over time (AAFC, 2016). We then cross-referenced the environmental externalities listed in this report with scholarly literature evaluating environmental externalities from Canada and from other countries (e.g. Jongeneel, Polman, and Van Kooten, 2016; Pretty et al., 2000; Tegtmeier and Duffy, 2004). Given the limited availability of studies evaluating externalities, we only make international comparisons with the U.K., the U.S. and the EU. This process yielded the following common externalities (in no particular order):

- GHG emissions;
- Ammonia (NH_3);
- Particulate Matter (PM) pollution;
- Nitrogen (N) water pollution;
- Phosphorous (P) water pollution;
- Pesticide water pollution;
- Coliform water pollution;
- Soil erosion;
- Wildlife and biodiversity;
- Wildlife habitat capacity (WHC); and
- Landscape aesthetics.

This list represents the most commonly studied environmental externalities from agricultural production, both positive and negative. We do recognize that other externalities exist, and we discuss a few of them in the section "Non-monetized impacts." Our report relies exclusively on secondary data, and as such we were unable to venture too far into unresearched territory. However, we believe this to be a relevant area for future research.

In the second step, we measured physical quantities of externalities by drawing from the aforementioned AAFC ag-environmental indicator report (AAFC, 2016). While we do not conduct the physical measurements ourselves, for each externality we describe the process by which AAFC developed each indicator. We analyze the physical measurement of agricultural GHG emissions in significant detail, using satellite data from Statistics Canada (2019). The satellite data provide information about land cover types across Canada at the field-level. When combined with the spatially-explicit GHG data from AAFC, we can derive a reasonable estimate of GHG emissions by crop type over location. And while GHG emissions are equally damaging wherever they occur, provincial policy-makers have made it clear that provincial-level emissions are important factors for public policy decisions, especially regarding carbon taxation.

In the final step, we estimate a value of the externalities using the value-transfer or benefit-transfer method. This method relies on existing non-market valuation studies, “transferring” their valuation estimates to our application. This method can perhaps best be explained by example. To value the damage created by GHG emissions, we rely on an estimate referred to as the social cost of carbon (SCC) which is an estimate of the damage done to society from emitting one tonne of CO_2-equivalent. The study we use estimates a SCC of C$39/tonne in 2012 dollars (Environment and Climate Change Canada (ECCC), 2016). To use the benefit-transfer method, we then apply this valuation to each tonne of GHGs emitted by province for each year of available data. In using the benefit-transfer method, we rely on studies that employ various methods of non-market valuation, most commonly the stated-preference approaches: contingent valuation and choice experiments. In stated-preference approaches, researchers rely on surveys and questionnaires to elicit personal valuations related to the externality. An example of a close-ended contingent valuation survey might ask someone if they would be willing to pay (WTP) $X to reduce phosphorous water pollution by 10%.

In addition to relying on studies that use stated-preference approaches, we use studies that attempt to calculate externalities in a more direct manner. For example, in the case of N water pollution, we use the cost of removing N from the water supply at a water treatment plant as an estimate for the cost of the externality. For externalities that have a direct impact on human health, like PM pollution, we rely on a study that uses a measure called the value of a statistical life (VSL), which places a dollar value on the years of life lost to high exposures to PM pollution.

For each externality, we use the benefit-transfer method in conjunction with the most appropriate study for our purposes. In the following sections and subsections, we describe the study used to value each externality. We do not show preferences for studies using one method over another (e.g. contingent valuation vs. choice experiment), but rather we select studies that are most relevant to Canadian agriculture. A promising avenue for future research would be to initiate the collection of primary data using a combination of stated-preference, revealed-preference, and direct estimation methods to better align our valuations with our Canadian-specific physical measurements. However, in the absence of primary data collection, we believe that the results presented in this report constitute the most accurate valuation currently available.

4 Results for Negative Externalities

Negative agricultural externalities consist of non-market effects deemed to have a negative effect on productivity, human health, and/or ecosystem health. In the agricultural sector, negative externalities include GHG emissions, air pollution, water pollution, soil erosion, and damage to wildlife habitat and biodiversity. Physical measurements and non-market valuations for each externality are enumerated in the following sections.

4.1 Greenhouse gas emissions

GHG emissions from agriculture consist mainly of carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4). Each of these gases have different global warming potential (GWP) per molecule. The GWP is the measure of the ability of a GHG to trap radiation and hence contribute to global warming (Marthin & Hoppe, 2016). CH_4 has 21 times the GWP of CO_2, and N_2O is 310 times the GWP of CO_2 when measured over 100 years (UNFCCC, 2018). Because of the differences in GWP per molecule, the acceptable standard of reporting is by CO_2 equivalence (CO_{2eq}) (Worth et al., 2016).

CO_2 emissions in agriculture arise mainly through the decomposition of crop residue and soil organic matter (SOM). CO_2 emissions from fossil fuel consumption in Canadian agriculture are not generally included in the calculation of agricultural GHG emissions because they are usually reported in the energy and transportation sectors to conform with international standards (Worth et al., 2016). Almost all CH_4 emissions from Canadian agriculture are from livestock production through enteric fermentation and anaerobic decomposition. More than 80% are directly from enteric fermentation in ruminants and the remaining are from livestock manure decomposition (Weersink & To, 2001). N_2O is emitted directly from field-applied fertilizers, crop residue decomposition, manure storage and cultivation of organic soils (Worth et al., 2016). N_2O can also be emitted indirectly through volatilization and N leaching and runoff.

4.1.1 Physical measurement

Agriculture, excluding emissions from fossil fuel use, contributes about 8.2% of total net GHG emissions in Canada (Prairie Climate Centre, 2018). AAFC tracks and calculates the net GHG emissions (emissions minus removals or sinks) from agriculture from 1981 to 2011 with their GHG indicator in the 2016 AAFC publication on Agri-Environmental Indicators. This indicator measures the net GHG emissions from the three primary GHGs associated with agriculture. In creating the indicator, a Canada-specific Intergovernmental Panel on Climate Change (IPCC) Tier II methodology is used to estimate the emissions. Estimating N_2O and CH_4 emissions involves the collection of primary input data, such as climate and farming systems; an estimation of interim variables such as N crop residue N; and finally, GHG emissions which are calculated by multiplying the amount of N (N_2O emissions) and the animal population (CH_4 emissions) by their emission coefficients (Worth et al., 2016). With CO_2, sophisticated computer models are used to estimate the emissions from cropland. Emissions are reported in units of CO_{2e}, which is consistent with the IPCC standard. Converting the primary GHG into CO_{2e} involves multiplying their masses by their respective GWP coefficients (i.e. $CO_2 = 1$; $CH_4 = 21$; and $N_2O = 310$).

4.1.2 Variation across crops and geography

GHG emissions from agriculture are driven by a number of factors, two of which are geography (i.e. variation in land characteristics across space) and crop selection. In the following five figures, Figure 4.1 through Figure 4.5, we describe differences in GHG emissions intensity by agricultural land use (crop selection) and province. These results closely mirror the previous figures showing variation in fertilizer emissions intensity, primarily due to the outsize effect of N fertilizer use on GHG emissions. For more information on how these estimates were generated, please refer to the Appendix.

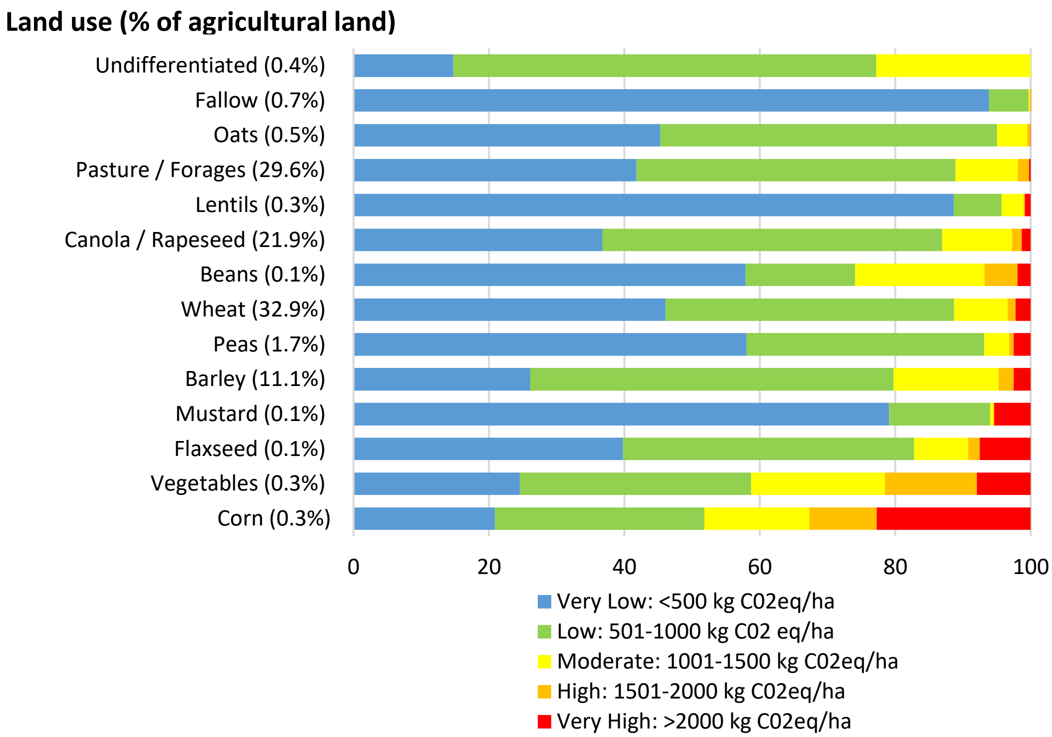

Figure 4.1. Relative Importance of GHG Emissions Intensity Categories (by hectare) for major agricultural land uses in Alberta – 2011

Source: AAFC and Statistics Canada

In Alberta, corn has the highest GHG emissions intensity, but was only planted on 0.3% of all agricultural land in the province. The two largest land use categories – wheat (32.9%) and pasture/forages (29.6%) – have the most acres in the very low and low GHG emission intensity categories, defined as less than 500 kg CO_2eq/ha and between 501 and 1000 kg of CO_2eq/ha. As a point of reference, a typical passenger car emits about 4,600 kg of CO_2 in a year.[12]

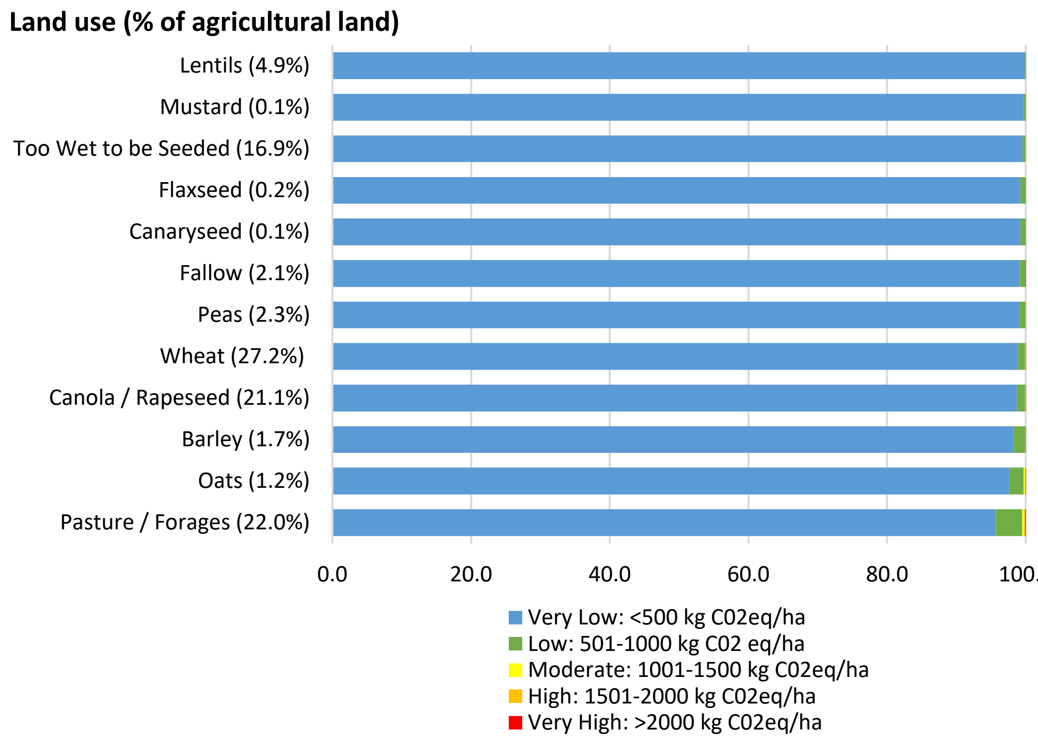

Figure 4.2. Relative Importance of GHG Emissions Intensity Categories (by hectare) for major agricultural land uses in Saskatchewan – 2011

Source: AAFC and Statistics Canada

The overwhelming majority of Saskatchewan's agricultural land falls in the lowest category of GHG emissions intensity. The relative difference between Saskatchewan and the more GHG intensive provinces like Quebec and Ontario (see Figure 4.4 and Figure 4.5) is striking. Saskatchewan's low GHG intensity is driven by the high adoption of conservation tillage, soil with high organic carbon, and crop selection that requires limited excess N (i.e. fertilizer).

[12] https://www.epa.gov/greenvehicles/greenhouse-gas-emissions-typical-passenger-vehicle

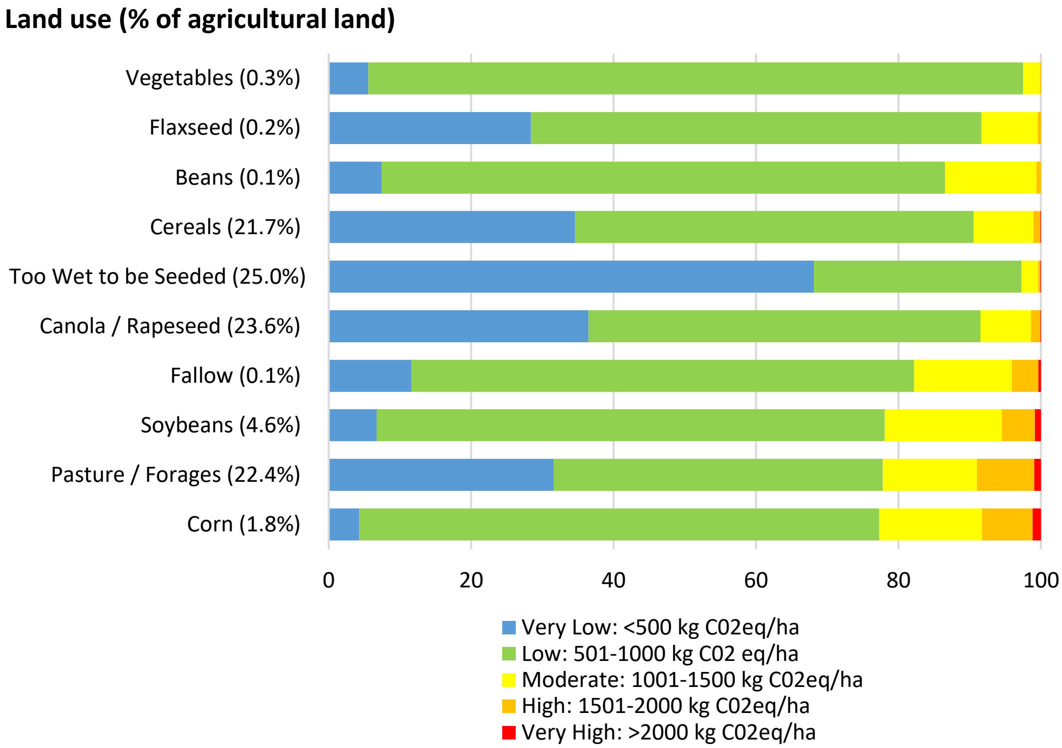

Figure 4.3. Relative Importance of GHG Emissions Intensity Categories (by hectare) for major agricultural land uses in Manitoba – 2011

Source: AAFC and Statistics Canada

GHG emissions intensity in Manitoba is also relatively low compared to Central and Eastern Canada. As is the case in Alberta, corn land contributes the most acres to the very high GHG emissions intensity category, but corn is only planted on 1.8% of agricultural land in Manitoba. As corn acres increase with varieties designed for the Manitoba climate, so could Manitoba's GHG emissions intensity.

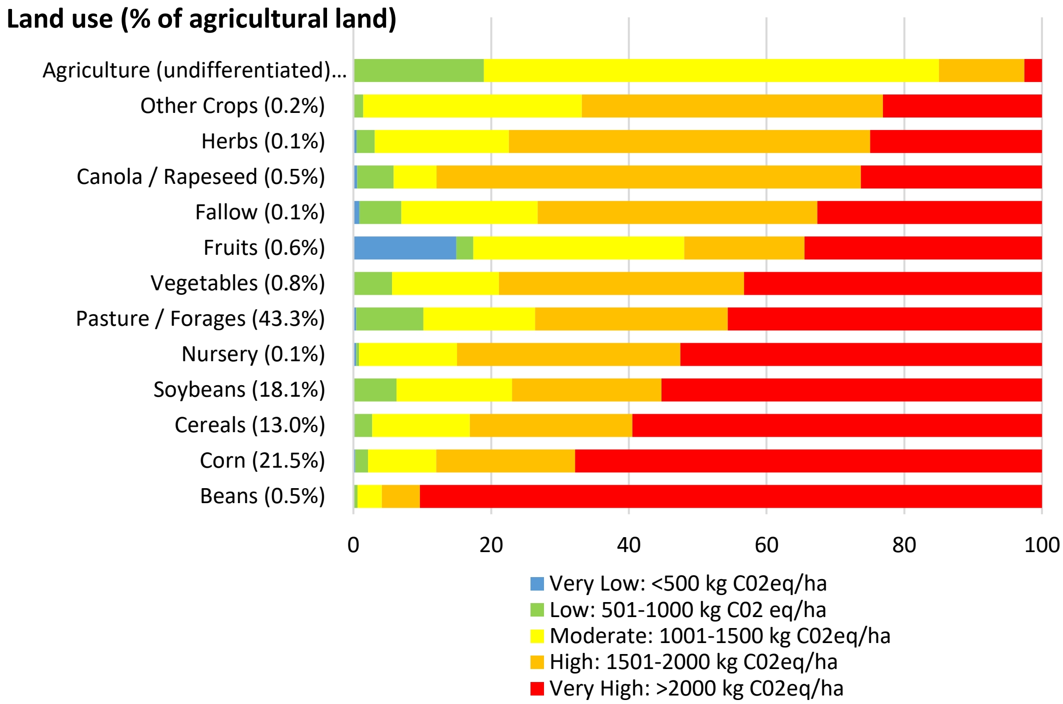

Figure 4.4. Relative Importance of GHG Emissions Intensity Categories (by hectare) for major agricultural land uses in Ontario – 2011

Source: AAFC and Statistics Canada

Agricultural land in the Central Canadian provinces of Ontario and Quebec is highly GHG intensive relative to their western counterparts. This is due primarily to differences in crop selection, along with soil types that are less amenable to conservation tillage. The largest contributor to GHG emissions in these provinces is corn, which is grown on 21.5% of Ontario's agricultural land (Figure 4.4) and 22.4% of Quebec's agricultural land (Figure 4.5).

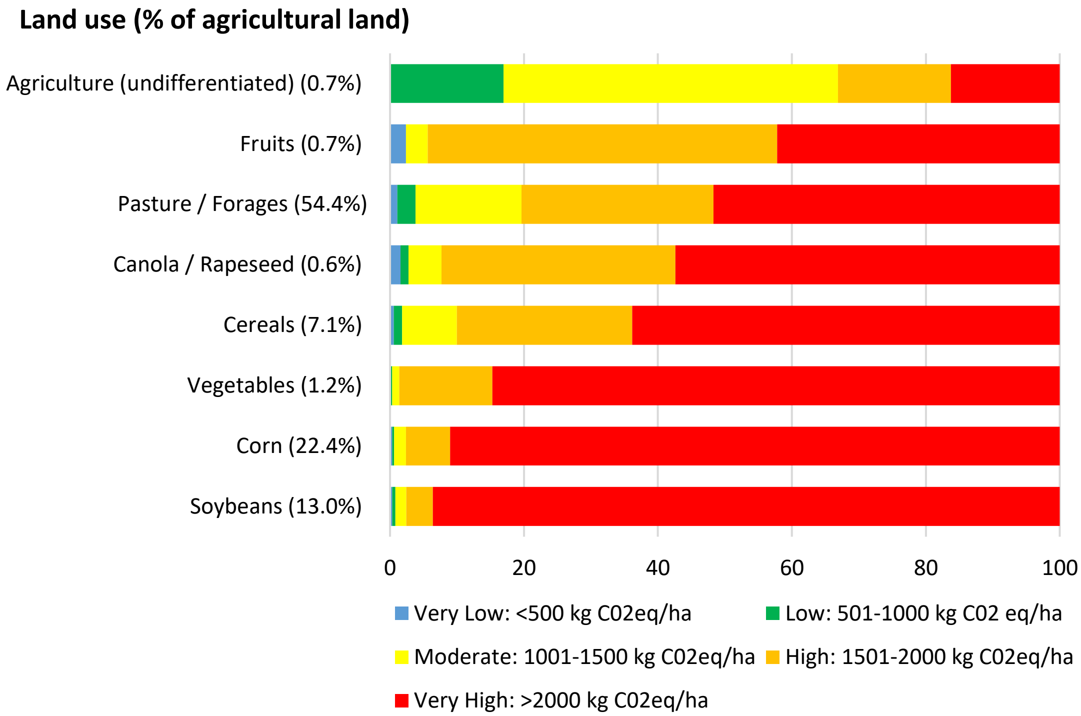

Figure 4.5. Relative Importance of GHG Emissions Intensity Categories (by hectare) for major agricultural land uses in Quebec – 2011

Source: AAFC and Statistics Canada

4.1.3 Valuation

To estimate the value of negative externalities arising from GHG emissions from agriculture, we use the social cost of carbon (SCC) concept. The SCC is the monetary value of the marginal damage from an additional unit (tonne) of CO_2 emitted into the atmosphere in a given year (Environment and Climate Change Canada (ECCC), 2016). ECCC endorses the use of the SCC to evaluate changes in CO_2 emissions in economic analysis, including in cost-benefit analyses for public policy decision-making (Dupras et al., 2016). In 2016, ECCC estimated an SCC of C$39/tonne (2012 prices). The SCC does not represent the price of C in Canada, but reflects the additional incremental damage expected from a small increase in CO_2 emissions (ECCC, 2016). To obtain the total external cost of GHG emissions, we multiplied the SCC by the total emissions over time. GHG emissions can be split between livestock and crop production based on their contribution to total emissions. Livestock contributes about 60% of GHG emissions from agriculture in Canada, and the remaining 40% can be attributed to crop production (ECCC, 2018).

Our approach is similar to Pretty et al. (2000) who use the marginal cost of various GHGs from previous studies in estimating the external cost of GHG emissions in U.S. agriculture. Using the SCC reflects the cost of GHG emissions on society, and hence is an accurate reflection of the impact, assuming emissions are accurately captured. In contrast, Tegtmeier and Duffy (2004) used the market price of GHG credits traded by the Chicago Climate Exchange, at $0.98/tonne in 2003, to estimate the external cost of GHG emissions from U.S. agriculture. This price only represents what companies are willing to pay to reduce GHG emissions but does not take health and environmental impacts into consideration. This is likely to result in the underestimation of the external cost of GHG emissions. Jongeneel et al. (2016) assume a shadow price of €16/tonne to estimate the external cost of GHG emissions in the Netherlands. This value is more of a market price since it is the average of traded annual prices. Hence, it is also likely to result in the underestimation of this cost. Therefore, we deemed the use of SCC as the most appropriate method.

4.1.4 Results

We present the external cost of GHG emissions in Table 4.1. All provinces included in the estimate exhibit a rising external cost for GHG emissions, except for Saskatchewan and Ontario. The total external cost in both regions together amounts to $1.68 billion in 1981 and $1.5 billion in 2011, representing a 10% reduction over the period. Most of the significant change occurred between 1991 and 2001 (Table 7.4).[13]

[13] For Table 7.4 see page 77 of this Report.

From 1981 to 2011, the external cost of GHG emissions in Saskatchewan decreased by 76% while that in Ontario fell by only 13% (Table 4.1). By comparison, the external cost of GHG emissions in Manitoba increased by 34.3% over this period. Alberta and Quebec both showed an increase in these costs of about 9%. Regionally, Western Canada had external costs from GHG emissions of $771 million in 2011, down 5% from $810 million in 1981. Central Canada, on the other hand, recorded external costs from GHG emissions of $479 million in 2011, up from $440 million in 1981 with Ontario accounting for $440 million and Quebec, $331 million in 2011.

TABLE 4.1. External costs (negative externalities) of Canadian GHG emissions, 1981 – 2011 (millions of 2012 dollars)

Province	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**Percentage change (1981-2011)**
AB	440.14	432.35	471.30	541.41	549.20	518.04	479.09	8.85
SK	292.13	257.07	237.60	257.07	144.12	105.17	70.11	-76.00
MB	136.33	155.80	159.70	202.54	202.54	214.23	183.07	34.29
ON	506.35	479.09	444.03	447.93	432.35	463.51	440.14	-13.08
QC	303.81	303.81	296.02	319.39	331.08	327.18	331.08	8.97
Region								
Western	810.16	782.90	740.05	767.32	763.42	790.69	771.21	-4.81
Central	440.14	432.35	471.30	541.41	549.20	518.04	479.09	8.85

4.1.5 International comparisons

An international comparison shows how other countries are doing relative to Canada in terms of external costs from GHG emissions. In the U.S., Tegtmeier and Duffy (2004) estimated a cost of $1.57/ha from GHG emissions. The same parameter for the five provinces in Canada in our report was estimated at about $28/ha. The higher value for Canada does not necessarily come from higher GHG emissions from Canadian agriculture, but from the estimation values employed in the study. In Canada's case, we used the SCC of about $40/tonne for valuing these costs, which is significantly higher than the market value of \$0.98/tonne used in the U.S. estimate. The estimate for the five provinces would fall to \$0.90/ha if the same value were used in our analysis as in the U.S. study after converting it to 2012 Canadian dollars.

The UK and the Netherlands had estimates that are above those for Canada. The UK study estimated external costs from GHG emissions per hectare of farmland at about \$215/ha, partly due to the high value of £63/tCO_2 used in the valuation and partly due to relatively more intensive agriculture in the UK. The average estimate for the Netherlands is about \$290/ha which can be attributed to the country's relatively intensive agriculture, with the value used of €16/tCO_2e. For instance, the Netherlands, with only about 1.9 million hectares of farmland, was the second highest exporter of agricultural products in the world in 2014, in value terms (Dillinger, 2017).

4.2 Ammonia

Ammonia gas (NH_3) is a natural product of microbial metabolism in livestock and livestock waste. It is released mainly through the breakdown of naturally excreted urea from cattle and pigs or uric acid from poultry. It can also be released from N fertilizer containing ammonium or urea. The majority of NH_3 emissions from agriculture come from livestock, although this is starting to change as livestock production becomes more efficient and declines in favour of crop production (Sheppard and Bittman, 2016).

At high concentrations, NH_3 can have noticeable impacts on human health, such as irritating the eyes and nose (New York Department of Health, 2004). In the atmosphere, it can also react with other acidic gases to form secondary PM matter less than 2.5µm in diameter. This can increase the incidence of respiratory diseases (New York Department of Health, 2004).

4.2.1 Physical measurement

Agriculture contributes about 85% of the total anthropogenic NH_3 in Canada (Ayres, Bittman, Sheppard, & Girdhar, 2010). According to AAFC (2016), 333,136 tonnes of NH_3 gas were emitted from agriculture in 1981, 420,866 tonnes in 2006, and 371,258 tonnes in 2011 (Sheppard and Bittman, 2016). In 1981, livestock contributed to about 81% of agricultural NH_3 emissions, and fertilizer contributed 19%. However, with the changing trend towards more crop production and less livestock, in 2011 livestock contributed to 65% of these emissions while fertilizer emissions accounted for 35% (Sheppard and Bittman, 2016). In 2011, beef production contributed about 47.7% of total emissions from livestock, and swine contributed about 24.6% (Figure 4.6).

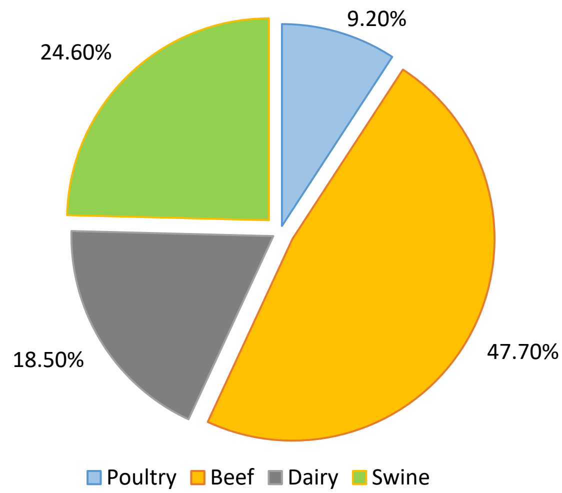

Figure 4.6. Share of NH_3 emissions by livestock type, 2011.

Source: Desjardins et al. (2016)

NH_3 emissions are estimated by the Ammonia Emissions from Agriculture Indicator by AAFC. The indicator is based on a computational model that uses data from several sources to estimate the annual emissions of NH_3 into the atmosphere from livestock production and fertilizer application. The models are modified for each subsector (broilers, turkeys, swine, calves, heifers, dry cows, lactating cows, cattle, beef cattle and fertilizer) to account for their particular attributes (Sheppard and Bittman, 2016). The source of data for the model includes information on farm practices in 12 ecoregions from farm surveys focusing on NH_3 emissions, Census of Agriculture data on livestock numbers, industry data on fertilizer use, and NH_3 emission factors adapted to Canadian farm practices and conditions.

4.2.2 Valuation

Muller and Mendelsohn (2007) estimate a value of $2,700/tonne as the marginal damage of NH_3 emissions in the U.S. They use the Air Pollution Emission Experiment and Policy analysis model (APEEP), an integrated assessment model, to calculate the marginal damage associated with emitting an additional tonne of a pollutant from about 10,000 sources in the U.S. The model is used to calculate the concentration of pollution, exposure, physical effects and dollar damages based on a baseline level of emissions. The model uses emission inputs to predict seasonal and annual average county-level concentrations of the air quality parameters. Exposure is then calculated by multiplying the concentration by the inventory of organisms and resources which could be damaged by these emissions. The exposure is translated into a physical effect using a concentration-response function. The concentration-response function establishes the relationship between exposure and the relative risk of some condition of a defined population. The physical effects are then converted to their dollar-equivalents. To convert the physical effects to dollar values, Muller and Mendelsohn (2007) use the VSL of $6.2 million. To calculate marginal damage, they estimate the total national damage, then the total damage after one tonne of a pollutant is added. The difference between the two results gives the marginal damage concerning the air pollutant under consideration.

We adopt the NH_3 marginal damage value of Muller and Mendelsohn (2007) because it is independent of the source of pollution, making it acceptable across sectors. We adjusted for population differences between the U.S. and Canada by weighting the Canadian population by a ratio of the Canadian to the U.S. population. The models also consider atmospheric chemistry which attributes all damages from secondary pollutants to the emissions that contributed to them. That is, damages from PM formed from NH_3 emissions are attributed to NH_3 gas. This prevents double counting of damages. We adjust their value using the 2002 purchasing power parity index to convert them to Canadian dollars (OECD, 2014) and to 2012 prices using the Canadian CPI (Bank of Canada, n.d.). This estimate, after accounting for the population difference, is $466.38/tonne from NH_3 emissions.

Our approach is similar to Pretty et al. (2000) who use a pre-existing value for NH_3. The value was estimated by identifying emissions, changes in exposure or impact, quantification of impact, and valuation based on a Willingness to Pay (WTP) value. The significant difference is that the WTP value in their study was calculated based on the value of a life year (VOLY) concept versus our estimate based on the VSL.

To estimate the total external cost of NH_3 emissions from agriculture, we multiply the value from (Muller & Mendelsohn, 2007) by the quantity of agricultural NH_3 emissions.

4.2.3 Results

The external costs of NH_3 emissions from agriculture in Western Canada and Central Canada are presented in Table 4.2. These costs in Western Canada (Alberta, Saskatchewan and Manitoba) increased by 43.7% between 1981 and 2011. However, the value for 2011 of $962 million represents a decline from the 2006 value of $1,087 million. Of the three provinces included in these regional estimates, Manitoba recorded the largest increase, of 57%. External costs from NH_3 emissions in Manitoba rose from $121 million in 1981 to $191 million in 2011. However, Alberta had the highest external cost at $305 million in 1981 and $416 million in 2011. The external costs of NH_3 emissions in Saskatchewan increased by 46% between 1981 and 2011, rising from $244 million to $356 million.

Between 2006 and 2011, emissions from livestock fell by 22% while emissions from fertilizer increased by 13%, with the majority of these changes observed in Western Canada (Sheppard & Bittman, 2016). The trend towards increased cropland and fewer livestock between 2006 and 2011 accounted for the decrease in NH_3 emissions and subsequently the decrease in external costs over this period.

Central Canada (Ontario and Quebec), on the other hand, recorded a decline in costs from NH_3 emissions of 17.4% between 1981 and 2011 when they fell from $650 million to $537 million respectively. Ontario recorded costs of $360 million in 1981 and $291 million in 2011, a reduction of about 19%. Quebec recorded external costs of $289 million in 1981 and $246 million in 2011, reflecting a reduction of about 15%.

In 1981, Ontario contributed the most to external costs from NH_3 emissions of the five provinces, with Alberta the second largest contributor. Manitoba contributed the least. In 2006 and 2011, Alberta was the largest source of costs from NH_3 emissions, probably due to the increase in livestock production in that province over this period. This was followed by Saskatchewan.

TABLE 4.2. External cost of NH_3 emissions, 1981 – 2011 (millions of 2012 dollars)

Province	**1981**	**2006**	**2011**	**Percentage change (1981-2011)**
AB	304.92	514.83	416.03	36.44
SK	243.65	372.62	355.69	45.98
MB	121.11	199.81	190.55	57.33
ON	360.49	334.82	290.59	-19.39
QC	289.25	273.61	246.13	-14.91
Region				
Western	669.69	1,087.26	962.28	43.69
Central	649.74	608.43	536.72	-17.40

4.2.4 International comparison

Considering that livestock contributes to NH_3 emissions globally, livestock-intensive countries are likely to have higher external costs from NH_3 emissions. The UK and Canada are similar in terms of the livestock and crops they produce. Both countries produce wheat, barley and canola as well as cattle, sheep, goats and poultry. Any differences in external costs should therefore come from the intensity of agricultural production. Because UK agriculture tends to be more intensive, we expect the UK will have higher negative externalities from NH_3 emissions compared to Canada.

The average external cost of NH_3 emissions in Canada for the five provinces is $2.70/ha, compared to the UK, at $9.73/ha. The WTP value of £171/tonne used by Pretty et al. (2000) was derived using the VOLY concept, whereas the value of $2700/tonne used by Muller and Mendelsohn (2007) employed the VSL. All things being equal, the Canadian estimate should be higher than the UK because of the difference between the two values used in the estimation process and due to the fact that UK agriculture is more intensive.

Similarly, the Netherlands has higher estimates for external costs from NH_3 than do Western or Central Canada. The estimate for the Netherlands stood at $281.46/ha. Jongeneel, Polman, and Cornelis Van Kooten (2016) use a restoration cost of €3.14/kg for the estimated 128.2 million kg of NH_3 from manure in the Netherlands.

About 54% of agricultural land in the Netherlands is grassland, which is important for their substantial dairy industry (Nations Encyclopedia, n.d.). Thus, even though the value used in evaluating NH_3 emissions is negligible compared to the value we use for Canada, the external cost per unit of agricultural land is higher in the Netherlands because the NH_3 emitted per hectare is greater than that of both Western and Central Canada.

4.3 Particulate matter

Particulate matter (PM), also known as particle pollution, is a mixture of solid particles and liquid droplets in the air. Agriculture is a significant contributor to particle pollution (Saxton, 1996) which comes from wind erosion, land preparation, crop harvesting, pollen, grain handling, crop residue burning, animal feeding operations, and fertilizer and chemical applications (Pattey et al., 2016). PM pollution can decrease visibility and reduce the amount of solar energy reaching the surface of the earth. It contributes to stratospheric ozone depletion, acid rain, and smog (Pattey et al., 2016). The resulting reduction in visibility can affect cities, airports, and wilderness areas, which adversely affects tourism and the economy. Fine particles also increase the incidence of respiratory diseases, such as asthma and chronic bronchitis, and premature deaths (Donham and Thelin, 2016; Samet and Krewski, 2007; U.S. EPA, 2004). It therefore can lead to increased emergency room visits and hospitalization.

Environmentally, the impact of PM emissions can cause changes to soil and water chemistry which can adversely impact vegetation and organisms. It can also stain buildings, including those of cultural importance such as statues and monuments (Canada Council of Ministers of the Environment (CCME), 2017).

4.3.1 Physical measurement

Agriculture contributed about 5% of primary PM emissions in Canada in 2006. AAFC estimates PM emissions from agriculture for the 1981 to 2011 period using the Agricultural Particulate Matter Emissions Indicator (APMEI). This indicator is based on a model developed to measure the emissions of primary PM from agricultural operations and to assess emission-reduction measures (Pattey et al., 2016). The model uses activity data for each agricultural source and emission factor to calculate the indicator. The emission factor is defined as “an estimate or statistical average of the rate at which a contaminant is released into the atmosphere through an activity, divided by the level of that activity” (Clearwater et al., 2016). The indicator estimates primary PM from wind erosion and from crop and livestock production activities that generate PM for the census years 1981 to 2011. More than 75% of all PM emissions from agricultural activities come from land preparation and wind erosion and about 90% of agricultural PM pollution is due to crop production (Pattey et al., 2016).

Total PM emissions from Canadian agriculture are classified into total suspended particles with an aerodynamic diameter less than 100µm (TSP), an aerodynamic diameter less than 10µm (PM_{10}), and an aerodynamic diameter of less than 2.5µm ($PM_{2.5}$). $PM_{2.5}$ and PM_{10} are the two classes that cause major health issues (Pattey et al., 2016). As such, we consider only these two latter classes in estimating external costs from PM.

The APMEI indicated improvement in particle pollution from agriculture in Canada over time (Figure 4.7). In 1981, the index indicated that particle pollution was at an 'at risk' level. This improved to a 'poor' level between 1986 and 2006. In 2011, the index improved to a 'moderate' level. This improvement can be attributed to the adoption of reduced tillage, no-tillage, and the decline in the use of summer fallow in the Prairie provinces (Pattey et al., 2016). The indicator varies by region since agricultural practices also vary by regions.

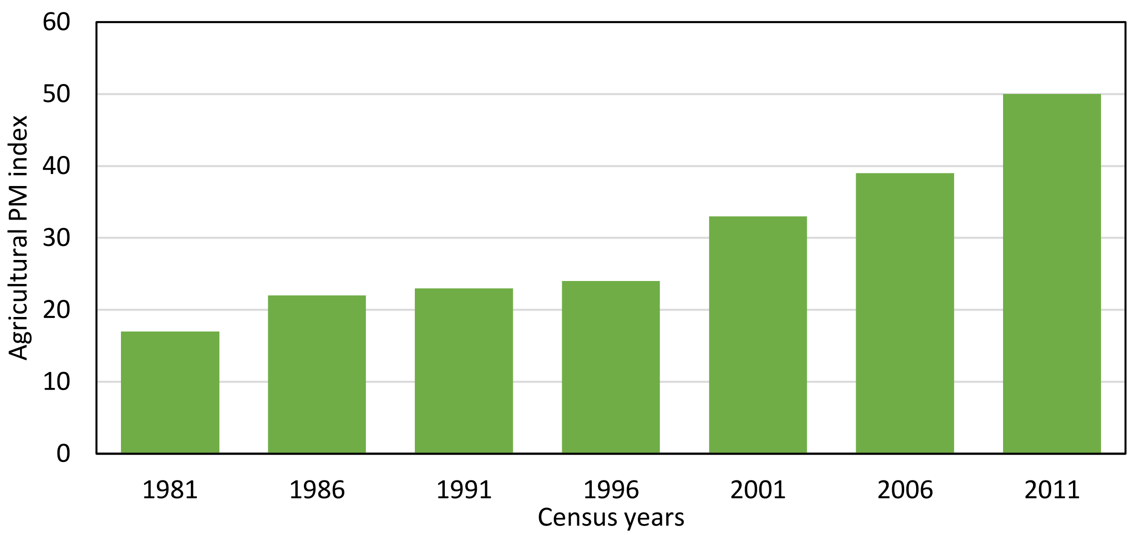

Figure 4.7. Agricultural PM index from 1981 to 2011 (higher index number means a smaller impact from PM pollution)

Source: Pattey et al. (2016)

4.3.2 Valuation

Muller and Mendelsohn (2007) estimated the marginal damage from PM_{10} emissions at \$350 (USD)/tonne and from $PM_{2.5}$ emissions, at \$2200/tonne for the U.S. The values converted to 2012 Canadian dollars are \$60.46/tonne for PM_{10} and \$380/tonne for $PM_{2.5}$ after adjusting for the population size difference in Canada versus the U.S. This approach was applied to NH_3 estimates as well and is described in more detail above in Section 4.2.2. The final step is to multiply the marginal damage of each PM class by its respective quantity emitted in order to estimate external costs of PM emissions in Canadian agriculture.

4.3.3 Results

Table 4.3 represents the external costs associated with agricultural PM emissions in Western and Central Canada. In 2011, these costs stood at $1.6 billion for both regions (Table 7.4). This represents a 60% reduction from the 1981 cost of $4 billion. Provinces in Western Canada reported the largest reduction in these costs between 1981 and 2011 (down 61%) but were also the provinces with the highest estimates (Table 4.3). Saskatchewan at $843 million and Alberta at $417 million accounted for more than 85% of the total costs from PM emissions in 2011. Quebec was the only province that recorded an increase in costs over the period,1981 to 2011, but these costs also were significantly lower than in the other provinces, at $49 million in 2011.

Saskatchewan reported the highest external costs due to PM emissions of $2.4 billion in 1981 and $843 million in 2011, representing a 64.5% reduction over the period. Alberta followed with a cost of $1 billion in 1981 and $542 million in 2011, a reduction of 60%. Manitoba reported the lowest costs in Western Canada, at $375 million in 1981 and $202 million in 2011, a 46% reduction. Total external costs from PM emissions in Western Canada were estimated at $3.8 billion in 1981 and $1.5 billion in 2011, a 61% reduction. This decline can be attributed to the adoption of zero and reduced tillage and the reduction in summer fallow. Also, in most areas, more tillage implies less straw management, which often involves burning. Some areas also passed legislation that makes burning of straw and stubble illegal.

Total external costs from PM emissions in Central Canada were estimated at $206 million in 1981 and $139 million in 2011. Ontario recorded a decline of 44% from $161 million in 1981 to $90 million in 2011. These costs increased in Quebec from $45 million in 1981 to $49 million in 2011, representing a 9% increase. While less amenable to no-till practices, Central Canada also saw farmers adopt new technologies and management practices that led to fewer PM emissions over this period.

TABLE 4.3. External cost of PM emissions, 1981 – 2011 (millions of 2012 dollars)

Province	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
AB	1036	995	905	828	646	542	417	-59.73
SK	2372	2116	1875	1689	1461	1131	843	-64.45
MB	375	340	305	293	270	234	202	-46.23
ON	161	155	149	130	114	105	90	-44.12
QC	45	45	45	46	53	49	49	8.93
Region								
Western	3,783	3,451	3,084	2,810	2,377	1,907	1,462	-61.35
Central	206	199	194	176	167	154	139	-32.48

4.3.4 International comparisons

We were unable to find studies from other countries with which to compare our estimates of external costs of PM pollution. Therefore, we are unable to compare this cost across jurisdictions.

4.4 Nitrogen (N) water pollution

N is vital for crop growth and productivity. However, excess application of N is not only economically costly for the producer but can also harm the environment. N gets into the soil through synthetic fertilizer application, organic manure application and natural atmospheric fixation by leguminous crops (Drury, Yang, De Jong, et al., 2016). Residual soil N (RSN), which measures the difference between the amount of N applied and the amount remaining in the soil after harvest, and other climatic factors influence the amount of N that ends up leaching into underground and surface waters (Drury, Yang, De Jong, et al., 2016). RSN can be exported from fields through leaching into water bodies and results in nitrate contamination of both ground and surface water (De Jong et al., 2009; Rochette et al., 2008). High levels of N can be harmful to aquatic life (i.e.

eutrophication) (Guy, 2008), and when found in drinking water, can lead to human health issues (Chambers et al., 2001). Eutrophication, the excessive growth of algae and aquatic vegetation in a body of water with a high concentration of plant nutrients, can lead to the depletion of dissolved oxygen in water, leading to the death of essential aquatic organisms (Clearwater et al., 2016).

4.4.1 Physical measurement

AAFC reported the N-loss from farmland in the 2016 Agri-Environmental Indicators report. The N-loss from farmland is reported in kilograms per hectare (kg/ha) at the provincial level for census years from 1981 to 2011. Because precipitation is a critical factor in transporting solution N, provinces with high precipitation have relatively higher rates of N loss per hectare compared to provinces with lower precipitation.

AAFC also created the Indicator of Risk of Water Contamination by N (IROWC-N) which is based on a model that establishes the link between the quantity of inorganic N remaining in the soil after harvest, or RSN, and subsequent climatic conditions during the winter (De Jong et al., 2009). The RSN is calculated as the difference between N inputs and N outputs. N is added through the addition of fertilizer and manure to farmland, leguminous fixation of N, and atmospheric wet and dry fixation of N. Outputs include N removal by harvested crops, gaseous loss of N into the atmosphere and N loss by leaching that was not captured by the RSN model. The model's main aim is to estimate nitrate-N leaching, which has the most harmful effects on the environment.

We use the annual N-loss rate reported for estimating external costs. Because Central Canada has high precipitation compared to Western Canada, the former region has a significantly higher risk of N-loss compared to Western Canada.

4.4.2 Valuation

Using a benefit-transfer approach, we adopted the estimates reported in Olewiler (2004). She estimated the cost of treating N at primary and secondary waste treatment plants in Vancouver in a range between C$3.04/kg and C$8.50/kg. The values were estimated based on self-monitoring and sampling data from the Greater Vancouver Regional District (GVRD) collected between 1 July 2001 and 30 June 2002. These prices are reported in 2012 Canadian dollars, at C$3.6/kg and C$10.1/kg. The midpoint of this range is C$6.86/kg. Wilson, (2014) also used this value as a proxy for the estimate of wetland waste treatment services for excess N. This value conforms best to the data available to us.

To calculate the total external cost of N water contamination, we multiplied the value (price) of N treatment ($/kg) by the total N emission rate (kg/ha) multiplied by total cropland area under production. We developed a low value, a moderate value, and a high value for this cost.

This approach is similar to previous work that estimated the external cost of N water pollution from agriculture. Jongeneel et al. (2016) used expenditures by water treatment companies in the Netherlands to remove N from water and the contribution of agriculture to N water pollution in the country to estimate the external cost of N water pollution. Pretty et al. (2000) used capital expenditures by water companies to remove nitrate from water and agriculture's share to estimate the negative impact of N pollution in the UK. Similarly, Tegtmeier and Duffy (2004) used the Environmental Protection Agency's (EPA) estimate of expected investment in infrastructure to meet the standards for surface water, coliform and nitrate pollution to estimate the total damages of nitrate water pollution in the U.S.

4.4.3 Results

The results presented in Table 4.4 show that Central Canada recorded the highest costs for N-water pollution from agriculture, while Western Canada showed the largest increase over time. Central Canada recorded costs of $663 million in 1981 and $796 million in 2011, an increase of 20%. Western Canada, on the other hand, recorded costs of $43 million in 1981 and $188 million in 2011, representing a 336% increase over the period.

Within Western Canada, costs of N water pollution rose significantly in Saskatchewan and Manitoba between 1981 and 2011. The costs in Saskatchewan jumped dramatically from $10 million in 1986 to $85 million in 2011. Saskatchewan reported zero costs in 1981 because the rate of N-loss for that period was nil, but costs rose to $85 million with the increase in fertilizer use for crop production (Drury, Yang, & De Jong, 2016). Manitoba recorded a total external cost for N water pollution of $13 million in 1981 and $75 million in 2011, an increase of 490%. Alberta saw a decrease in external costs in 2011 over 1981, falling from $30 million in 1981 to $28 million in 2011. This was after an increase to $42 million in 2006.

Central Canada recorded high external costs from N water pollution over the census years, primarily due to relatively high precipitation in the region. Cropping patterns and crop choices also play a significant role in the amount of N fertilizer used. For instance, Central Canada grows more corn than Western Canada and corn requires significant N fertilizer. N fertilizer usage per hectare and emissions intensity in Central Canada are consequently higher than in Western Canada (Table 2.1). The external cost of N water pollution in Ontario in 2011 was estimated at $368 million (Table 4.4). This is a decline of 14.5% from the 1981 value of $430 million. Quebec recorded total external cost of N water pollution of $233 million in 1981 and $429 million in 2011, representing an increase of 84% over the period.

Total external costs for both Central and Western Canada increased between 1981 and 2011. These costs were $706 million in 1981 rising to $985 million by 2011 (Table 7.4). This increase reflects expanding crop production and cropland and increasing N fertilizer usage over this period.

TABLE 4.4. External costs of N water pollution, 1981 – 2011 (millions of 2012 dollars)

Province	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
AB	30	33	27	41	35	42	28	-7.56
SK	0	10	0	10	22	43	85	784.23
MB	13	26	17	31	75	68	75	490.29
ON	430	501	475	424	427	428	368	-14.47
QC	233	287	287	303	383	400	429	83.93
Region								
Western	43	69	44	82	132	153	188	336.18
Central	663	788	762	728	810	828	796	20.13

4.4.4 International comparisons

When making a comparison internationally, the average external cost of N water pollution in Western and Central Canada was estimated at $16/ha, substantially higher than the U.S. ($0.7/ha) and the UK ($3.26) but lower than for the Netherlands ($102). Considering that corn is an important economic crop in parts of the U.S. and requires high N inputs, it is surprising to see this relatively low value for N water pollution in the U.S.

One explanation could be that the U.S. and UK values are estimated using projected capital investments by the government to reduce N water pollution and water companies' capital and operating expenditure respectively. These proxies may not be the best for estimating this cost. Our estimates for Canada and that of Jongeneel et al. (2016) for the Netherlands are comparable taking into account the quantity of N emitted from agricultural lands and the cost incurred by water treatment facilities to treat a kilogram of N. The difference in these two approaches could account for the difference between the estimates. There may be other factors that account for these differences as well.

The two major factors that are likely to influence the values are the average annual precipitation and the choice of crops grown, which influence the quantity of N added as fertilizer. Nitrate water pollution is influenced concurrently by residual soil nitrogen (RSN) and precipitation. Assuming Canada and the Netherlands have comparable RSN, the country with the higher precipitation will have a higher risk of N water pollution. Similarly, the country with the higher RSN will have a higher risk of N water pollution, especially if it has similar precipitation. In 2014, the average precipitation in the Netherlands was 778mm/year whereas that of Canada was 537mm/year (TheGlobalEconomy.com, n.d.).

Secondly, the crop choice of each country could also influence the amount of RSN which would subsequently impact N water pollution. The Netherlands grows sugar beets and potatoes, rye, and flower bulbs in addition to the main crops grown in Canada (wheat and barley). Potatoes and beets are generally known to require more N and P for growth. Therefore, the increase in production of these crops in the Netherlands could have led to the higher levels of N water pollution and subsequently higher external costs.

4.5 Phosphorous (P) water pollution

Phosphorous (P) is a nutrient needed by both plants and animals. It is an essential component of the energy-storing molecule Adenosine 5'-triphosphate (ATP), and plays a vital role in cell development (Science Learning Hub, 2018). It is applied to soil in the form of inorganic fertilizers, manure and bio-solids to maintain crop yields. Over the years, inputs of P as fertilizer or manure in Canadian agriculture has exceeded the amount taken up by plants resulting in cumulative P surpluses (Reid et al., 2016). This has increased the risk of soil P leaching from agricultural fields to surface water.

Excess P in water contributes to eutrophication (richness in dissolved nutrients in water bodies which stimulates the growth of aquatic plant life, resulting in the depletion of oxygen) and Cyanobacteria blooms, resulting in deteriorating water quality. Consequently, this can lead to the restricted use of water for drinking and recreational activities in some areas (Carpenter et al., 1998). This phenomenon is most evident in shallow lakes with a large proportion of their watershed under agriculture or urban land, like Lake Winnipeg in Manitoba, Lake Erie in Ontario and Missisquoi Bay in Lake Champlain (Reid et al., 2016).

4.5.1 Physical measurement

AAFC created the risk of water contamination by P indicator (IROWC-P) to assess the risk associated with leaching from agricultural sources of P. It is not clear what is the contribution of agriculture to P water pollution, but two factors concurrently determine the risk of P loss from soil, namely, P source, and P transport (Reid, Western, et al., 2016). These conditions vary across regions in Canada. As such, given the same level of P across regions, the Prairies will record a lower risk compared to Central or Eastern Canada, due to lower rates of P transport (Reid, Western, et al., 2016). Therefore, we believe that any measure of damage from P water contamination across the country should reflect the differences in the risk levels. However, the highest class level, which is 'very low' risk of P pollution has a P concentration of less than 1ppm (<1 mg P/L) (Reid, Western, et al., 2016). This concentration is above the eutrophication guidelines of 0.01mg P/L (Weersink & To, 2001). Therefore, our estimate of the external costs of P water contamination is the same for all provinces (regions). Also, apart from the 'very low' and 'low' risk levels, there are no significant differences in the shares of farmland under each risk category, specifically between the Prairie provinces and Central Canada. The share of farmland in the Prairies that falls under the 'very low' risk category is significantly higher than that of Central Canada (p-value = 0.0593). However, the share of farmland in Central Canada under the 'low-risk' category is also significantly higher than that of the Prairies (p-value = 0.0037). However, it is worth noting that the Prairies have about 18.4 million hectares of farmland under this category compared to the 4.5 million hectares in Central Canada.

4.5.2 Valuation

To estimate the total external cost of P water contamination in the Netherlands Jongeneel et al. (2016) used a benefit-transfer approach and used the unit value of

€10.3/kg from a previous study for the value of P water pollution. They multiplied this value by the quantity of P transported into water bodies from agriculture in the Netherlands to come up with the external cost of P water pollution. Pretty et al. (2000), on the other hand, used capital expenditures and operating costs of removing P and soil from water to estimate the total external cost of P and soil in water from agriculture in the UK. The issue is that this approach hinges on data availability.

Due to the lack of data on the quantity of P transported from agricultural land in Canada, we cannot use the unit price per kg of P in our estimation. We, therefore, used a WTP estimate from Larue et al. (2017) which estimated a WTP for a 10% reduction in P in water bodies in Quebec at C$0.65/acre ($1.60/ha) (2012 prices). The study used a stated choice experiment which took account of the risk behaviour of the respondents in the design and analysis of the experiment and results. Respondents were asked how much farmers should be charged to cover the cost of a 10% reduction in P water pollution. Their results showed that respondents were willing to impose a cost of $1.60/ha on farmers to achieve the 10% reduction in P water pollution. We consider this value to be the actual value of the perceived damage from P water pollution.

4.5.3 Results

table 4.5 shows our provincial estimates. In general, the external cost of P water pollution in both Western and Central Canada has increased over time. Manitoba and Ontario recorded a slight decline in costs over the period. In Western Canada, the total cost was estimated to be $39.4 million in 1981 and $46 million in 2011, representing an increase of 17% over the 1981 value. Saskatchewan had the highest costs in the region of $19 million in 1981 and $24 million in 2011, a 25.5% increase. Saskatchewan was followed by Alberta, in terms of value with external costs of $14 million in 1981 and $16 million in 2011. Manitoba recorded the lowest costs in Western Canada at $7.07 million in 1981 and $6.96 million in 2011, relatively unchanged over the period. These estimates are directly proportional to the area of farmland in each province, especially since area was used to estimate the results.

External costs of P water pollution in Central Canada have remained relatively stable over the census years, with about a 2% increase from 1981 to 2011. Ontario recorded costs of about $5.81 million in 1981 and $5.78 in 2011, relatively unchanged. Quebec, the province with the lowest costs, recorded costs of $2.81 million in 1981 and $3 million in 2011, an increase of 6.76%. The amount of cropland in each Province is a key factor determining these external costs.

TABLE 4.5. External cost of P water pollution, 1981 – 2011 (millions of 2012 dollars)

Province	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**Percentage change (1981-2011)**
AB	13.51	14.66	14.87	15.27	15.57	15.39	15.61	15.55
SK	18.79	21.32	21.53	23.04	24.60	23.94	23.57	25.45
MB	7.07	7.23	7.62	7.52	7.54	7.52	6.96	-1.62
ON	5.81	5.53	5.46	5.67	5.85	5.86	5.78	-0.52
QC	2.81	2.79	2.62	2.78	2.96	3.09	3.00	6.76
Region								
Western	39.36	43.21	44.02	45.83	47.71	46.85	46.13	17.19
Central	8.62	8.32	8.08	8.45	8.81	8.95	8.78	1.85

4.5.4 International comparisons

The external cost of P water pollution from agriculture in Canada (Western and Central) was estimated at 0.90/ha, while estimates for the UK were $11.14/ha, and the Netherlands, $30/ha. The likely reason for the difference in these values are similar to that of N water pollution. The UK used capital expenditures by water treatment facilities to derive their value, which makes it difficult to draw comparisons across the two countries. In addition, we expect more P to be transported from farms in the Netherlands than from Canadian farms, due to the higher precipitation in the Netherlands and more intensive farming. Finally, our cost estimate is derived differently from a WTP value from a Canadian study, based on a 10% reduction in P water pollution. We feel that the estimates for the

Netherlands are likely to capture the external cost of P water pollution from agriculture more accurately.

4.6 Pesticide water pollution

Pesticides are used in agriculture to reduce losses caused by weeds, insects, and plant diseases. These pesticides can also endanger human health through direct exposure, release into the environment (air and water), and residues in food (Tegtmeier and Duffy, 2004). Most pesticides are formulated to target a specific pest organism. However, some active ingredients in pesticides can be harmful to non-target species when pesticides move from on-farm application sites to contaminate the surrounding environment (Gagnon et al., 2016).

For example, pesticides from agriculture can enter surface and ground water through runoff and leaching (Tegtmeier and Duffy, 2004). Canada has experienced a rise in pesticide usage, particularly on the Prairies, with a 7% and 43% increase in herbicide and insecticide use, respectively, between 2006 and 2011. This was mainly due to the need for weed control with greater adoption of conservation tillage practices (Gagnon et al., 2016). Pesticide residues ranging between 0.5% to 5% of the amount applied have been found in surface and ground water in monitoring studies conducted in Canada (Cessna et al., 2005). In 2016, about 120.1 million kgs of active ingredients of pesticides were sold in Canada, of which about 75% were sold to the agricultural sector (Health Canada, 2017).

4.6.1 Physical measurement

Estimating the external cost of pesticide run-off from agricultural land requires data on the quantity of pesticides transported into water bodies and value estimates for a unit of a pesticide’s active ingredient. Unfortunately, such values are hardly available if they exist at all. AAFC developed the Indicator of the Risk of Water Contamination by Pesticides (IROWC-Pest) to assess the relative risk of pesticide contamination across agricultural areas in Canada (Gagnon et al., 2016). The indicator uses the Pesticide Root Zone Model to estimate the quantity of pesticides finding their way into the surrounding environment from pesticides transported into water bodies. The model uses data on agricultural practices, the quantity of pesticides applied to crops, and national pesticide use.

AAFC classifies the volume of pesticides transported into water and their concentration rates into five levels. By matching the volume of pesticides used with the share of farmland in each risk category would have been the most appropriate way to generate these values. However, we could not find these values. What we used instead was measured on a household basis, sourced by province from Canada Mortgage and Housing Corporation (CMHC) to quantify the damage or cost of pesticide water pollution.

4.6.2 Valuation

Tegtmeier and Duffy (2004) used the expenditures required for treatment facilities to meet regulations for pesticides under the Safety Drinking Water Act (SDWA) to approximate the external costs from agricultural pesticide use in the U.S. They used the share of pesticides sold to the agricultural sector to adjust for the total external cost for pesticide water contamination. Similarly, Pretty et al. (2000) also used annual capital expenditures by water treatment facilities on pesticide removal as a proxy for this cost. Using these approaches omits the cost of pesticides that are not covered under pesticide regulations for drinking water and might capture costs associated with non-agricultural-sourced pesticides as well. Jongeneel et al. (2016) used the cost to agricultural producers of reducing pesticide use to estimate the total external costs of avoiding water contamination from pesticides in the Dutch agricultural sector. This only reflects avoidance costs and not the total external costs from pesticide water pollution. To address this, we recommend using an estimate that reflects total environmental damage.

Brethoura and Weersink (2001) estimated the environmental benefits from pesticide reduction between 1983 and 1998 (15 years) at $187.61 per household in Ontario. This represents the amount each household would be WTP annually to experience a one percentage change in environmental risk over the period. However, this value includes other parameters such as human health benefits and wildlife. After adjusting for water-related estimates, this value fell to $ 46.55 per household (in 1993 US dollars). We adjusted this value using the Purchasing

Power Parity (PPP) indicator and the Canadian CPI to convert it to 2012 Canadian dollars with a value of $80.31 (Bank of Canada, n.d.; OECD, 2014).

4.6.3 Results

We estimate the external cost of pesticide water contamination over the period 1981 to 2011 for the five provinces in Western and Central Canada (Table 4.6). All provinces recorded an increase in their external costs from pesticide water pollution over this period. The external cost rose from $539 million in 1981 to $869 in 2011, representing a 61% increase in cost (Table 7.4).

In Western Canada, these externality costs were estimated to be $118 million in 1981 and $188 million in 2011, a 59% increase in cost over the period (Table 4.6). Alberta recorded the highest costs and the largest increase in the region with costs of $62 million in 1981 and $116 million in 2011, representing an 86% increase. Saskatchewan reported a cost of $27 million in 1981 and $34 million in 2011, representing a 26% increase. Manitoba's costs were estimated at $29 million in 1981 and $38 million in 2011, a 32% increase over the period.

Central Canada recorded relatively high costs from pesticide water contamination, at $421 million in 1981 and $681 million in 2011, a 62% increase. Ontario with the highest cost overall, recorded costs of $243 million in 1981 and $404 million in 2011, a 66% increase. Quebec recorded very high external costs from pesticide pollution at $177 million in 1981 and $277 million in 2011 (Table 4.6).

Total costs for pesticide water contamination showed a rising trend over the period with most of these costs occurring in Central Canada. These high costs can be partly explained by the higher precipitation rates in the region, which influence pesticide runoff from the application site into bodies of water. However, the greater population density and growing number of households in this region can also be a key factor contributing to these rising costs. This will have to be offset by the development of new pesticides with less harmful active ingredients that will have less harmful effects in the future.

TABLE 4.6. Externality cost of pesticide water pollution, 1981 – 2011 (millions of 2012 dollars)

Province	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
AB	62	69	74	81	91	105	116	86.23
SK	27	29	30	31	31	32	34	26.27
MB	29	31	33	34	36	37	38	31.58
ON	243	267	301	325	352	380	404	66.01
QC	177	194	217	230	244	259	277	56.22
Region								
Western	118	130	137	146	158	174	188	59.20
Central	421	462	518	555	596	639	681	61.88

4.6.4 International comparisons

According to the Food and Agricultural Organization of the United Nations (FAO), average pesticide use per hectare of cropland in 2012 for Canada was 1.53kg/ha, while in the U.S. it was 2.59kg/ha, in the UK, 2.83kg/ha, and in the Netherlands, 10.81kg/ha. To put this into perspective, the pesticide usage on cropland in Canada in 2012 was about 59%, 54%, and 14% of pesticide usage in the U.S., the UK, and the Netherlands respectively. These are consistent with our estimated average external costs of $14.22/ha for Canada, $24.29/ha for the UK and $100.57/ha for the Netherlands. In the U.S., however, this estimate was only $0.4/ha. These differences can be explained by the fact that for the UK, Pretty et al. (2000) use operating costs of £119.6 million/year for removing pesticides from water as the estimate for external costs. On the other hand, Jongeneel, Polman, and Cornelis Van Kooten (2016) use the cost of reducing pesticide in water of €143.7 million/year on the other hand. Therefore, the intensive use of pesticides in these countries is likely to lead to the difference in estimates compared to Canada. Intensive use of pesticides is also likely to stem from the agricultural practices adopted in each country as well as crop choices. The valuation procedures used could also account for the differences in the average values.

4.7 Coliform water pollution

Coliforms are thermo-tolerant bacteria found in animal faeces. They are not likely to cause illness, but their presence in drinking water indicates that pathogens may be present in the water system (Washington State Department of Health, 2016). Bacteria are an essential component of livestock manure (which is a vital source of nutrients for crop growth) that aid in the deposition of manure. The microbial composition in manure varies depending on the type of livestock and the health status of the livestock (Reid, Jamieson, et al., 2016). Therefore, animal manure used as a fertilizer input for crop production may pose a risk to the environment and human health if bacteria in the manure, and any associated bacteria, ends up in surface and shallow groundwater. Coliform contamination could increase the cost of water treatment, result in the loss of use of recreational waters with potential human health effects (Reid, Jamieson, et al., 2016). Canadians have become increasingly concerned about the quality of water they use for everyday activities such as drinking, washing and bathing (Martin, Clearwater, and Hoppe, 2016).

Areas with large livestock production, high manure production, dense water drainage networks and high susceptibility to surface runoff, preferential flow and erosion are likely to lead to the highest risks of surface water contamination by coliforms (Reid, Jamieson, et al., 2016). Consolidation of livestock farms with a reduction in the number of farms but an increase in farm size can lead to excessive production of manure that is above the capacity that surrounding farms can use. The high cost of transporting manure to cropland where it is used as fertilizer results in land in close proximity to livestock production often having higher levels of manure applied. This can lead to a higher risk of potential contamination of surface water by coliforms. It is for this reason that there are provincial regulations governing nutrient management and farm sizein many provinces (e.g. Alberta, Manitoba, Ontario, Quebec).

4.7.1 Physical measurement

The Indicator of the Risk of Water Contamination by Coliform (IROWC-Coliform) created by AAFC, assesses the relative risk of contamination of surface water bodies by faecal materials from agriculture (Reid, Jamieson, et al., 2016). This indicator uses thermos-tolerant coliforms as a marker and sources of coliform and their movement as data to determine the risk level. AAFC classified the risk into five levels. This makes it difficult for us to find numbers that can help in estimating the external costs from coliform pollution.

Instead, we use area in pastureland in each province over the 1981 to 2011 period to quantify the impact of coliform water contamination. Our decision is based on the fact that pastureland is mainly for feeding livestock or for livestock grazing, and coliform pollution from agriculture comes from this source.

The limitation of using pastureland to quantify coliform water pollution is that it might not be able to capture the pollution from confined animals, such as hogs. Also, manure spread on cropland might also be omitted in the estimation since some of the manure could come from confined livestock such as beef (feedlots), dairy, hogs, and fowl. Our approach might also result in an underestimation if croplands are used to produce livestock feed. Nevertheless, using pastureland is the most appropriate way to quantify the impact of coliform water pollution because grazed livestock accounts the largest source of coliform bacteria in both Eastern and Western Canada, with runoffs from pasture accounting for about 90% of the risk of coliform water pollution (Reid, Jamieson, et al., 2016).

4.7.2 Valuation

Previous studies that measure the external costs from coliform and pathogen water contamination use legal compliance costs as a proxy. Pretty et al. (2000) used expenditures on removing Cryptosporidium from water as a measure of the external cost of Cryptosporidium from agriculture in the UK. Similarly, Tegtmeier and Duffy (2004) used the EPA's national cost for implementing the Interim Enhanced Surface Water Treatment rule as a measure of external costs from agricultural pathogen water pollution. This approach reflects the cost to institutions responsible for water quality standards. However, the individual consumer cost due to pathogen water pollution from agriculture might not be reflected in these estimates. The consumer's WTP to reduce this damage indicates how much the consumer values the damage or the marginal cost to the consumer. Therefore, we recommend and used a WTP value as a measure of damage from pathogen water pollution.

Larue et al. (2017) estimated the WTP for a 10% reduction in coliform pollution for non-farm residents at C$2.27/ha in Quebec. This study used a stated choice experiment and accounted for risk behaviour of the respondents in the design and analysis of WTP values. Their results showed that respondents were willing to impose a cost of $2.27/ha on farmers to achieve the 10% reduction in coliform water pollution. This value could be close to the actual value that will be transferred to society due to coliform water pollution from farmlands. It should be noted that the 10% coliform reduction targeted in the Larue et al. (2017) study may not reduce the coliform level in water to the acceptable standard for drinking water. However, considering that not all coliform in water is from agricultural fields (Tegtmeier and Duffy, 2004), this value can be used to estimate a conservative cost that can be attributed to agricultural activities.

4.7.3 Results

External costs for coliform water pollution are higher in Western Canada than in Central Canada but declined in both regions between 1981 and 2011 (Table 4.7). Total external costs in both regions were $43 million in 1981, peaking at $44 million in 2006 and falling to $42 million in 2011, which is a reduction of about 2.7% from 1981 values (Table 7.4).

In Western Canada, the total external cost for coliform water pollution was $38 million in 1981 and $40 million in 2011, representing a 4.8% increase (Table 4.7). Provincially, Alberta reported the highest costs from coliform pollution with a cost of $17.4 million in 1981 and $20 million in 2011, showing a 15% increase. Saskatchewan had the second highest costs in Western Canada, at $15.9 million in 1981 and $15.86 million in 2011. Manitoba had costs estimated at approximately $5 million in 1981 and $4.3 million in 2011.

Central Canada, on the other hand, reported costs of $5 million in 1981 and $2.12 million in 2011, representing a 58% reduction over the period. For Ontario, these costs declined from $3.3 million in 1981 to $1.5 million in 2011, a 54% reduction. Quebec experienced the greatest improvement in external costs from $1.8 million in 1981 to $0.61 million in 2011, representing a 66% reduction.

Western Canada contributed most to these external costs. Alberta and Saskatchewan contributed to 75% of the total cost over the period, probably due to the large amount of pastureland in these provinces.

TABLE 4.7. External cost of coliform water pollution, 1981 – 2011 (millions of 2012 dollars)

Province	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
AB	17.35	17.82	18.90	19.57	20.09	20.59	19.93	14.85
SK	15.90	14.49	14.64	14.47	14.91	15.94	15.86	-0.28
MB	5.01	4.57	4.73	4.56	4.49	4.73	4.30	-14.14
ON	3.29	2.44	2.35	2.30	1.86	1.71	1.51	-54.00
QC	1.80	1.40	1.48	1.18	0.85	0.71	0.61	-66.34
Region								
Western	38.27	36.88	38.27	38.61	39.48	41.26	40.09	4.77
Central	5.09	3.84	3.83	3.47	2.71	2.42	2.12	-58.36

4.7.4 International comparisons

We compare our estimates for external costs for coliform water contamination in Canada to those of Pretty et al. (2000) for the UK. The average cost per hectare in Western and Central Canada stands at $0.69/ha while that in the UK was $5.85/ha. The difference can be attributed to the different estimation procedure used. While the UK estimate uses legal compliance costs for valuing external costs, we use WTP for a 10% reduction in coliform and cropland. It is unclear which country would have a higher average value if the same estimation procedures were used.

4.8 Soil erosion

Soil erosion occurs when the topsoil is removed from one location and deposited at another location. It is mainly caused by wind and water, but its occurrence is also influenced by tillage activities (Lobb et al., 2016). Wind and water erosion is also affected by certain agricultural activities and practices such as cultivation, summer fallow, and leaving the land bare after harvest (Tegtmeier and Duffy, 2004).

Soil nutrients, which are mainly found in the topsoils and subsoils, may be carried away when erosion occurs. This affects soil fertility, soil organic matter content, soil water holding capacity, and soil productivity (Tegtmeier and Duffy, 2004). To compensate for the loss in soil nutrients, farmers may use more fertilizers to increase crop yield. This has the potential of increasing the level of soil nutrients that can end up in water bodies. Apart from this indirect effect of erosion, off-site costs occur when wind or water-eroded soils increase the risk of floods, block roads and ditches, and damage properties (Belcher, Kallio Edwards, and Gray, 2001; Pretty et al., 2000).

Externalities, by definition, are costs imposed off-site. The costs related to decreased soil function on the farm are considered internal costs. When soil fertility falls as a result of soil erosion or degradation, the farmer either spends more money on fertilizer to replenish the lost nutrients or experiences reductions in yields. Either way, the result is a loss to the farmer. The off-site damages from soil erosion and the costs associated with them, however, are not necessarily taken into account by the farmer. Individuals, companies, and governments incur the following costs: the cost to water companies for removing soil sediments from water; the cost to the government for reconstruction of damaged roads and water conveyance infrastructure; the cost to individuals for renovating erosion-damaged buildings; and the cost to individuals, governments, or companies for restoring ditches, and water storage and drainage facilities.

4.8.1 Physical measurement

AAFC (2016) uses the risk of soil erosion index to ascertain the combined effect of wind, water and tillage erosion on cultivated agricultural lands. The indicator is derived as a function of climate, soil and topography and changes in farming practices over the census years 1981 to 2011 (Lobb et al., 2016). The index shows an upward trend in the risk of soil erosion from 1981 to 2011 which points to a reduction in the risk of soil erosion. Over this period, the index improved from 65 to 84 on a 100 scale, indicating about a 28% improvement overall. The improvement is mainly due to the adoption of reduced tillage and no-till and the decline in summer fallow particularly on the Prairies.

Depending on soil type and its characteristics, the rate of erosion may vary by different soil zones and regions. AAFC classified these soil zones into five classes, with different erosion risks, namely, 'very low risk' (<6t/ha/year), 'low risk' (6-11t/ha/year), 'moderate risk' (11-22t/ha/year), 'high risk' (22-33t/ha/year), and 'very high risk' (>33t/ha/year). To quantify the soil erosion occurring on farmland, we calculated the proportion of cropland in each risk level using the percentages provided by AAFC. The sum of the product of the cropland in each of the risk levels with their respective erosion rates yielded the total soil eroded. We use the minimum values for 'very low risk' and 'very high-risk' levels as the erosion rate and use the midpoint value for the remaining levels. That is, we used erosion rates of 6t/ha/year for 'very low risk', 8.6t/ha/year for 'low risk', 16.5t/ha/year for 'moderate', 27.5t/ha/year for' high', and 33t/ha/year for 'very high'.

4.8.2 Valuation

To estimate the external cost from soil damages from agriculture in the Netherlands, Jongeneel et al. (2016) considered CO_2 emissions (i.e. GHG) from peat soils, the cost for water management, the cost associated with soil organic matter loss and erosion and compaction by using benefit-transfer values. We did not use their approach because 1) our estimates for GHG emissions are net of carbon sinks as described in Section 4.1.1, and 2) we deemed the costs associated with on-farm water management as a private cost not to be included in external cost estimates. Soil erosion and compaction costs also include the costs associated with leaching of N into water bodies and N_2O emissions in the air. Adding these to our estimates would result in double counting.

Tegtmeier and Duffy (2004) use different subcategories to estimate the total cost from damage to soil resources. This includes the cost to the water utilities industry, the lost capacity of reservoirs, the cost to water conveyance systems, flood damages, the cost to recreational activities, the cost to navigation, and other in-stream and out-stream costs. Their estimates are mainly based on restoration and replacement costs.

Similar to Jongeneel et al. (2016), Pretty et al. (2000) also use off-site damage caused by soil erosion and organic matter and CO_2 losses to estimate the total external costs from soil erosion. However, their estimates for off-site damage was based on the cost to local authorities arising from damage to properties and roads but did not include the cost to water companies.

A combination of these approaches would have been the preferred approach. However, the lack of data availability in our studies does not permit us to use any of these approaches. This notwithstanding, our approach has a higher potential for estimating more accurate external costs from soil erosion in agriculture. The available data from AAFC segregates the proportion of land under different risk levels and does not assume erosion occurs uniformly across croplands. This is consistent with the work of Graves et al. (2015) who quantified soil erosion damage by considering different erosion rates across varying landscapes. What the previous studies' estimates imply are that all agricultural lands will yield the same level of damage. Our estimate can capture the difference in the rate of erosion that may arise due to the difference in soil type, type of vegetation cover, and the topography of the land.

Pimentel et al. (1995) estimated off-site costs of erosion in the U.S. at $17 billion (1992 USD). They also stated that 4×10^9 tons of soil are lost through erosion from cropland annually. From these values, we calculated a value of $7.68/tonne (2012 CAD) for the off-site damage. This includes damage from both wind and water erosion. We multiply this value by the area (hectares) of cropland in each risk category and also by their respective erosion rates to come up with the external costs for soil erosion.

4.8.3 Results

The estimated external costs of soil erosion on Canadian farmland are presented in Table 4.8. Generally, external costs from soil erosion have been declining over time. The total costs for Western and Central Canada declined by about 28% from $2.84 billion in 1981 to $2.05 billion in 2011. The highest cost of $2.95 billion was reported in 1986. Since then external costs from soil erosion have plummeted.

Western Canada reported a 32% reduction in the cost of soil erosion over the period (Table 4.8). A total cost of $2.23 billion was estimated for 1981 and $1.5 billion in 2011 (Table 7.4). The reduction in external costs for soil erosion can be attributed to the adoption of reduced and zero tillage and a reduction in summer fallow in the region. About 50% of this cost comes from Saskatchewan, 30% from Alberta, and 20% from Manitoba. In Saskatchewan, external costs of soil erosion in 1981 were $1.17 billion declining to $768 million in 2011, a reduction of 34.6% (Table 4.8). External costs in Alberta were estimated at $728 million in 1981 and $497 million in 2011, representing a 32% reduction. Manitoba reported external costs of $328 million in 1981 and $249 million in 2011, showing a 24% reduction.

Only Quebec, among the five provinces studied, exhibits a slight increase in costs from soil erosion. External costs rose from $111 million in 1981 to $112 in 2011, representing a 0.6% increase. In Ontario, external costs of soil erosion fell from $502 million in 1981 to $424 million in 2011, representing a 16% reduction. For the region as a whole, external costs of soil erosion were $613 million in 1981 and $535 million in 2011, also due to an increase in conservation tillage.

TABLE 4.8. External costs of soil erosion, 1981 – 2011 (millions of 2012 dollars)

Province	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
AB	728	779	745	678	626	540	497	-31.75
SK	1174	1247	1215	1182	1119	868	768	-34.58
MB	328	332	300	302	308	282	249	-24.01
ON	502	484	470	465	465	421	424	-15.63
QC	111	108	98	106	118	116	112	0.59
Region								
Western	2,230	2,357	2,260	2,162	2,054	1,689	1,514	-32.11
Central	613	593	568	571	583	537	535	-12.70

4.8.4 International comparisons

With more than 50% of its agricultural land in grassland, the Netherlands has the lowest average external costs from soil erosion at $2.10/ha. Since these grasslands are tilled less or not tilled at all, soil erosion on such lands will be minimal. Also, the Netherlands produces many crops in greenhouses, eliminating the possibility of erosion.

The estimate for the U.S. is an on-farm estimate and hence cannot be considered in this comparison. The UK estimate, on the other hand, uses replacement cost as a proxy for the external cost of soil erosion and hence may not be comparable as it does not fully reflect the actual external cost.

4.9 Wildlife and biodiversity

Agricultural land provides habitat and a breeding ground for wildlife. It also provides wildlife a source of food and protection. However, modern agriculture has had a profound effect on wildlife. The 579 identified species of birds, mammals, reptiles, and amphibians that use the Canadian agricultural landscape for breeding, feeding and shelter require unique habitats to maintain viable population (Javorek et al., 2017). Wildlife habitats have been degraded through the intensification of agriculture since it has encouraged the conversion of natural and semi-natural lands into cropland. It has also led to increased livestock intensity, the increased use of chemical inputs, the removal of wetlands, shelterbelts and natural field barriers in order to accommodate large machinery. This has resulted in habitat fragmentation and loss of landscape heterogeneity (Javorek et al., 2017).

Wildlife habitat fragmentation usually results in a reduction in wildlife habitat capacity (WHC). That is, people lose the utility or benefit that they derive from activities that rely on wildlife and biodiversity, such as bird watching or hunting because smaller portions of habitat usually support smaller and less diverse wildlife populations.

4.9.1 Physical measurement

A recent World Wildlife Fund (WWF) report states that there was a 83% decline in half of the monitored wildlife species in Canada (451 out of 903) from 1970 to 2014 (WWF, 2017). The primary threat to wildlife species in Canada is habitat loss from agriculture, urbanization, forestry and industrial development. Though climate change, pollution, and invasive species are also reported as threats (World Wildlife Fund, 2017).

Agriculture's contribution to WHC loss is not explicitly known. The natural habitat within the agricultural landscape is estimated to support 89% of wildlife species associated with farmland (Javorek et al., 2016). On the other hand, cropland together with unimproved pasture, woodlands and wetlands and other land covers (e.g. farm buildings, lanes, idle lands, barns) can only support 49% of these species (Javorek et al., 2016). This means 40% of the remaining species may be lost due to agricultural activities.

AAFC created the WHC on Farmland Indicator (WHCFI) to assess the broad-scale trends on the Canadian agricultural landscape in providing suitable habitat for populations of terrestrial vertebrates (Javorek et al., 2016). The indicator was constructed by obtaining land-cover information at the Soil Landscapes of Canada

(SLC) polygon level from census data, and a habitat associated matrix was constructed and linked to a group of 579 species of interest, i.e. deemed valuable to the local ecosystem. The trend over time was determined through analysis of variance, followed by a comparison of the means of WHC to determine significant changes in WHC. Changes are classified as 'large decrease', 'small decrease', 'constant', 'small increase', and 'large increase', with the proportion of farmland that falls under each category. We use the proportion of farmland in the 'small decrease' and 'large decrease' categories to quantify WHC loss to estimate the external costs from agriculture's impact on wildlife and biodiversity. We ignore the 'constant' category since it represents farmland experiencing no change in WHC over the 1981 to 2011 period.

4.9.2 Valuation

Belcher et al. (2001) estimate the WTP for wildlife improvement at \$52 (\$17.52 – consumptive (e.g. hunting); \$34.49 – non-consumptive (e.g. bird-watching)) from a conservation cover program in the Grand River watershed in Southern Ontario and a WTP for wildlife improvement of \$15 (\$10.71 – consumptive; 4.16 for non-consumptive) from a conservation cover program in the Upper Assiniboine River Basin in Eastern Saskatchewan and Western Manitoba.[14] In estimating the consumptive WTP value, they use the expenditure on hunting wildlife and the consumer surplus estimate from Environment Canada (2000) for one day, translated into a per hectare value.

To estimate total external costs for wildlife and biodiversity, we use the estimates from Belcher et al. (2001). We adjust the values to 2012 constant prices using the Canadian CPI. To address the difference in WHC between the 'large decrease' and the 'small decrease', we propose the use of 50% of the value applied to the 'small decrease' while the original benefit-transfer value is applied to the 'large decrease'.

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[14] We freely admit that the values in Belcher et al. (2001) are themselves benefit-transfer calculations—which may result in a loss of accuracy when extended again to our context. These findings should be interpreted only as a rough approximation conditional on the limitations of the available data.

Pretty et al. (2000) use the cost of restoring wildlife species and habitats to measure the external cost of agriculture's impact on wildlife and biodiversity in the UK. Tegtmeier and Duffy (2004) use benefit-transfer values based on the valuation of the environmental impacts of pesticides by Pimentel et al., 1992. What is unique about our estimation approach is the ability to address the heterogeneity of agriculture's impact on wildlife and biodiversity across space by differentiating the effects on farmland.

4.9.3 Results

The external costs of wildlife and biodiversity loss in Western and Central Canada are presented in Table 4.9. These costs have fallen dramatically over time. The total cost in Canada was $286 million in 1981 and $253 million in 2011, representing a 12% reduction (Table 7.4). This reduction suggests that Canada has made efforts to reduce the threats to wildlife species and their habitats in promoting sustainable agriculture over this period.

In Western Canada, total costs from wildlife loss were $47 million in 1981, rising to $49 million in the 1986 to 2006 period (Table 4.9). Alberta reported costs of $19.5 million in 1981, rising to $21.1 million in 2011, up 8.6%. Saskatchewan, on the other hand, experienced a decline in these costs of about 3.7%. The province reported a cost of $20 million in 1981 and $19.2 million in 2011. Manitoba registered the lowest cost in the region, at $7.3 million in 1981 and $6.9 million in 2011.

Most of the costs associated with wildlife and biodiversity loss due to agriculture is reported in Central Canada. Total costs of $239 million were recorded in 1981, falling to $205 million in 2011, a decline of 14.3%. In Ontario, the external costs of wildlife loss were $184.5 million in 1981 falling to $156 million in 2011, a 15% decline. Quebec showed a smaller decline of about 11.7% as external costs fell from $55 million in 1981 to $48.6 million in 2011.

TABLE 4.9. Cost of wildlife and biodiversity loss, 1981 – 2011 (millions of 2012 dollars)

Province	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
AB	19.45	20.79	21.18	21.52	21.68	21.83	21.12	8.61
SK	19.96	20.36	20.57	20.34	20.14	20.00	19.23	-3.68
MB	7.27	7.35	7.35	7.34	7.22	7.34	6.93	-4.70
ON	184.49	172.49	166.53	171.58	166.98	164.54	156.69	-15.07
QC	54.98	52.94	49.90	50.28	49.71	50.38	48.58	-11.65
Region								
Western	47	49	49	49	49	49	47	1.28
Central	239	225	216	222	217	215	205	-14.28

5 Positive externalities

Positive externalities from agriculture are also important. They consist of non-market goods or services deemed to have a positive effect on productivity, human health, and or ecosystem health. In our study, positive externalities are estimated for the provision of wildlife habitat and landscape aesthetics. Physical measurements and non-market valuations for each externality are enumerated in the following sections. Carbon sequestration is a positive externality arising from agricultural practices, but at present it is not a net positive externality as is the case with wildlife habitat benefits and landscape aesthetics.

5.1 Wildlife habitat

Agriculture can lead to benefits for wildlife through the provision of a complex landscape and habitat that is essential for certain species of wildlife (Javorek et al., 2016). As discussed earlier, agriculture is an important cause of habitat loss for certain species of wildlife, imposing costs on decreased wildlife and biodiversity. However, for other species of wildlife, the agricultural landscape can provide shelter, habitat and food among other amenities. Different species of wildlife have different habitat requirements for breeding, shelter and feeding. The availability of these unique habitats is essential to maintain viable wildlife populations. Canadian agricultural land includes cultivated lands, hay lands and grazing lands, as well as semi-natural cover types such as wetlands, woodlands, riparian areas and grasslands (Dupras et al., 2016). Some agricultural management practices can enhance habitat for certain wildlife species, including rotational grazing, shelterbelts, winter cover crops, buffer zones around water bodies and tillage practices that retain surface crop residues (Statistics Canada, 2015). This diversity in land use cover addresses the different habitat requirements of different species of wildlife and has been recognized to improve WHC on some farmland in Canada (Javorek et al., 2016). Improvements in WHC yields both consumptive and non-consumptive benefits to Canadians (Belcher et al., 2001). Well-managed native pasture and rangelands are valuable for wildlife. Other species that also benefit from agricultural habitat include white-tailed deer, coyotes, snow geese, and moose. For example, Fox and Abraham (2017) reported that in contrast to the declining populations of most North American bird species, northern hemisphere geese, including snow geese and Canada geese have increased in abundance due in part to increases in food quality and availability from corn, cereals, legumes and winter green cereals. Laforge et al (2016) argue that land use change in agricultural regions of Saskatchewan, such as crop fields interspersed with tree, shrub and wetland cover types combined with lower predation risks in agricultural landscapes have resulted in more favourable conditions for moose. It is also understood that activities related to agriculture, including expansion of agricultural crops, winter feeding of cattle and the reduction of competitors and predators have contributed to the expansion of white-tail deer populations (Walter et al, 2009)

5.1.1 Physical measurement

The 2016 AAFC Agri-Environmental Indicators report provides data on the share of farmland in Canada that has increased its WHC for select species of wildlife. Based on this information the Canadian agricultural landscape demonstrated improvements in WHC over the period being studied. The proportion of farmland that improved varied over time and across provinces. Between 1996 and 2011, Canadian agriculture has maintained a constant WHC on about 82% of farmland across the country (Javorek et al., 2016). We use the proportion of farmland that showed improvement in WHC to quantify the benefits to wildlife. The improvement in WHC was classified into ‘small’ changes and ‘large’ changes. AAFC used the percentage of land corresponding to each land-cover type to determine the species-specific habitat availability (SSHA) for a given Soil Landscape of Canada (SLC) polygon. For each SLC polygon, a single value for potential WHC on farmland was created by taking the average of all SSHA values calculated for that polygon. A large increase represents a potential WHC value greater than 76 and a small increase represents a potential WHC value between 59.6 and 76. To quantify the externality, we multiply the agricultural land in each year by the proportion of land in each WHC change category. Even though we only have these figures for 1996 to 2011, we assumed the same change for 1981.

Figure 5.1 and Figure 5.2 show WHC on farmland in Western and Eastern Canada. Category 1, indicated by the orange area, shows the highest relative WHC, and Category 5, indicated by the violet area, shows the lowest WHC. Western Canada has more agricultural land with high WHC, which is a function of the larger farm area, larger farm sizes and much lower population densities.

Wildlife Habitat Capacity on Farmland in Western Canada (2011)

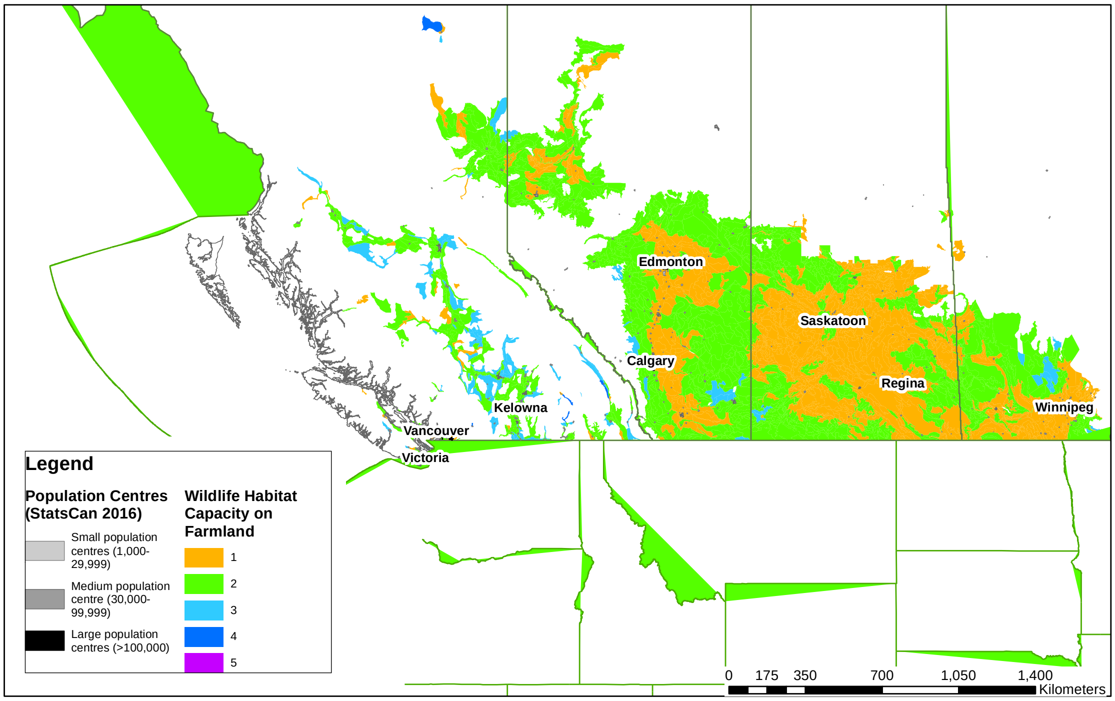

Figure 5.1. Spatial relationship between wildlife habitat and farmland in Western Canada

Source: AAFC and Statistics Canada

Wildlife Habitat Capacity on Farmland in Central and Eastern Canada (2011)

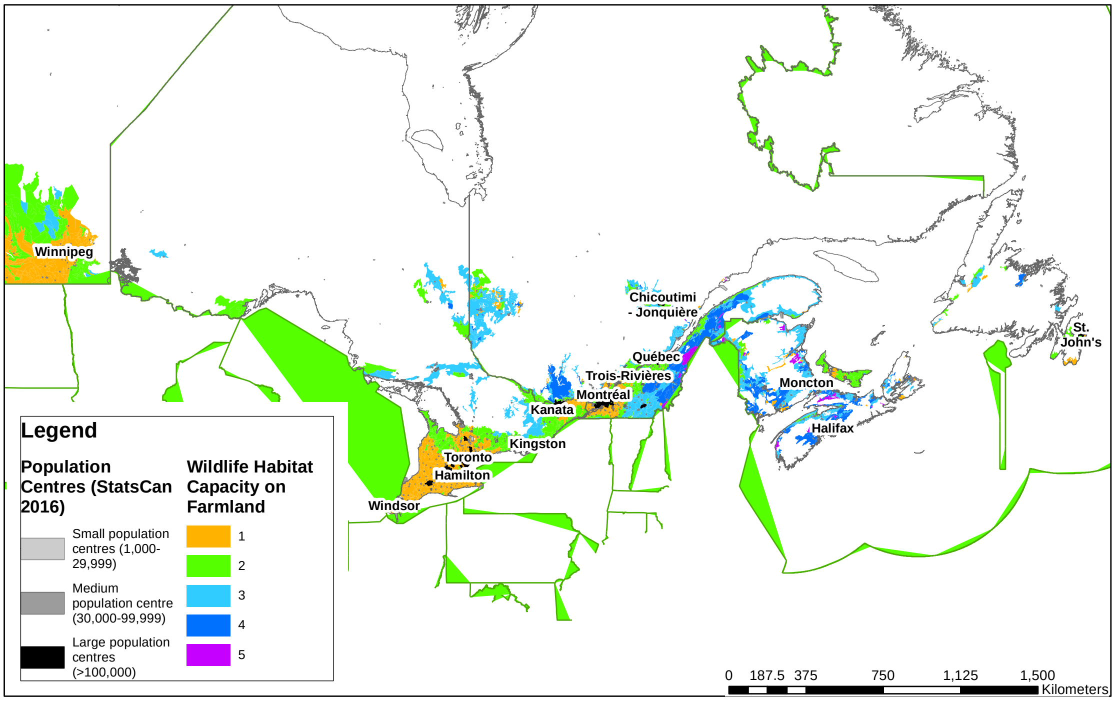

Figure 5.2. Spatial relationship between wildlife habitat and farmland in Eastern Canada

Source: AAFC and Statistics Canada

5.1.2 Valuation

The economic value of biodiversity and wildlife habitat is usually measured based on its uses for society, even though nature has an intrinsic value which is arguably greater (Kulshreshtha and Kort, 2009). Dupras et al. (2016) estimated the unit value of biodiversity habitat for the Gatineau Park in Quebec in 2011 at $2,196/ha/year in Canadian dollars. Even though this value is not too large from the eco-centric perspective (intrinsic value of nature), we found that it was too large compared to other values in the literature. Kulshreshtha and Kort (2003) use a range of values between \$249/ha and \$361/ha to estimate the value of enhanced biodiversity from shelterbelt in the Prairies. These values are adjusted from Scott et al. (1998) who estimated the value of biodiversity as the cost of establishing groundcover with natural vegetation. Establishing groundcover, though necessary for soil microbial habitat, could serve other purposes such as erosion control. It may also not reflect a complete habitat for wildlife. Therefore, the estimate is likely to be an overestimate of the correct value.

Using the benefit-transfer technique, we rely on the values from Belcher et al. (2001) to value the benefits from the provision or improvement of wildlife habitat.[15] They estimate a value of \$15/ha as both consumptive and non-consumptive wildlife benefits of a conservation vegetation program in the Upper Assiniboine River Basin in Eastern Saskatchewan and Western Manitoba and \$52/ha in the Grand River watershed in Southern Ontario. We use the Saskatchewan and Manitoba value for Western Canada and the Ontario value for Central Canada. We multiply these values by the proportion of agricultural land in each change category to get the external benefits from wildlife.

5.1.3 Results

Table 5.1 presents the external benefits from wildlife habitat improvement due to agricultural activity in Western and Central Canada over the period of analysis. Generally, these benefits have been decreasing over time, which is consistent with the WHC on Farmland Index developed by AAFC (Clearwater et al., 2016). The indicator showed a decline from a score of 40 in 1981 to 36 in 2011. Our estimates valued this benefit at \$38 million in 1981 and \$32 million in 2011 (Table 7.4).

These benefits declined in Western Canada by 3% from a value of \$3.75 million in 1981 to \$3.64 million in 2011 (Table 5.1). Alberta, the only province showing some improvement, saw benefits rise from \$0.37 million in 1981 to \$0.4 million in 2011, representing a 6.84% increase. In Saskatchewan, benefits from wildlife

[15] An important caveat for using this estimate is the original author's use of permanent cover crops in their estimation. Extending this interpretation to the entire agricultural landscape is inexact at best.

habitat improvement ranged from $3.01 million in 1981 to $2.9 million in 2011, a decline of 3.8%. Manitoba's estimates ranged from $0.37 million in 1981 to $0.35 million in 2011.

For Central Canada as a whole, benefits fell from $34 million in 1981 to $29 million in 2011. Ontario had the highest benefits over the period, but also the sharpest decline of 15% between 1981 and 2011. Estimates for the province were $33.42 million in 1981 and $28.39 million in 2011.[16] Quebec, on the other hand, reported much smaller benefits, which also declined from $0.51 million in 1981 to $0.45 million in 2011.

TABLE 5.1. External benefits to changes in wildlife habitat, 1981 – 2011 (millions of 2012 dollars)

Province	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
AB	0.37	0.40	0.40	0.41	0.41	0.41	0.40	6.84
SK	3.01	3.09	3.12	3.08	3.05	3.02	2.90	-3.84
MB	0.37	0.37	0.37	0.37	0.37	0.37	0.35	-4.23
ON	33.42	31.25	30.17	31.08	30.25	29.81	28.39	-15.07
QC	0.51	0.49	0.46	0.46	0.46	0.46	0.45	-11.65
Region								
Western	3.75	3.86	3.90	3.87	3.82	3.80	3.64	-2.82
Central	33.93	31.74	30.63	31.55	30.71	30.27	28.83	-15.02

5.2 Landscape aesthetics

Landscape aesthetics can provide significant social benefits associated with agricultural landscapes. Maintaining and adding value to the agricultural landscape involves the interaction between humanity and the natural landscape

[16] Ontario estimates appear out of line since they are 100 higher than the other provinces. However, this is based on the value ($52/ha) taken from the study of the Grand River Conservation area in Ontario, which was higher than the value used on the Prairies ($15/ha).

(Brady, 2006). The rural landscape, which comprises both natural and anthropogenic features, is transformed through producers' decisions regarding land use, output and input levels. Maintaining rural landscapes is considered a desirable outcome among OECD countries and country policies often reward them (Weersink and To, 2001). The benefits derived may include open views, crop diversity, pleasing pastoral landscapes, architectural elements, and personal attributes (i.e. emotional attachment and family heritage) (Dupras, Laurent-Lucchetti, Revéret, & Dasilva, 2018).

5.2.1 Physical measurement

Quantifying the benefits from landscape aesthetics is a controversial and challenging task. This is partly because aesthetics become somewhat subjective with one's views of beauty and attractiveness. Some people may prefer that the agricultural landscape remains in its natural state. Such people will value the altered-agricultural landscape less than those who believe that a tilled landscape is more attractive than a natural one. The aesthetic value placed on grassland and tamed and seeded pastureland may also vary from the aesthetic value of cropland and from one person to another. Agricultural land with farm buildings and cropland might have different values and this can also be a challenge in quantifying the benefits of landscape aesthetics. The location and topography of the agricultural landscape can also influence the value people place on its aesthetics. In summary, different land use within the agricultural landscape will have different aesthetic values and will also vary across individuals and across space. This makes it difficult for deciding how to quantify landscape aesthetic benefits.

We use all agricultural land in each of the five provinces for this estimate in this report. However, results from this quantification approach need to be considered with caution for the reasons provided in the preceding paragraph and also because most of the values used are non-use values which have little or no market transactions associated with them to provide a realistic context.

5.2.2 Valuation

The economic value of landscape aesthetics is usually affected by the landscape type and the people who value it (Dupras et al., 2016). Because of this, the most popular approach to valuing landscape aesthetics is through surveys (Dupras et al., 2018; Junge et al., 2015; Fleischer and Tsur, 2009; Brady, 2006). Dupras et al. (2016) estimated a value of $76/ha for cropland aesthetics, $176/ha for grassland and pasture, and $4/ha for freshwater systems for the Gatineau Park in Quebec. These values were based on the mean estimate derived from a benefit-transfer method. We used the benefit-transfer value of $76/ha in our estimate since the majority of agricultural land is cropland. We adjusted this value to 2012 Canadian dollars using the Canadian CPI.

Of all the valuations in this report, these values are perhaps the most fraught with issues. There is no agreement in the literature about the correct method of valuing landscape aesthetics due to the extremely high degree of subjectivity among those living in the landscape and those that do not. An infamous example of concessions made to this second group is the case of pastoral farmers in Switzerland being publicly subsidized to graze ruminants in the mountains (Schulz, Lauber, and Herzog, 2018). It is unclear how such a scheme could (or would) be implemented in Canada.

5.2.3 Results

Once again, the values presented in Table 5.2 should be viewed with caution because they do not take account of the variations across space and jurisdiction. The only variation these estimates capture is the difference in area of agricultural land. The external benefits from agricultural landscapes in Western Canada and Central Canada were $4.6 billion in 1981. The benefits increased from 1981 to 1996 when they peaked at $4.75 billion. They then declined from 1996 to 2011, when they reached $4.5 billion, a decline of 2.2% over the entire period.

Landscape benefits in Western Canada were higher mainly because most Canadian farmland is in this region (Table 7.2). Total benefits for landscape aesthetics in this region were $3.9 billion in 2011, unchanged from 1981. As expected, Saskatchewan reported the highest benefit values over the period followed by Alberta and Manitoba. Saskatchewan's landscape benefits were estimated at $1.9 billion in 1981 and $1.84 billion in 2011. Alberta is the only province whose estimates rose over the period from $1.41 billion in 1981 to $1.5 billion in 2011, representing an increase of 6.8%. Manitoba's estimates were the lowest in the region with a benefit of $561 million in 1981 and $538 million.

Total external benefits from agricultural landscape aesthetics in Central Canada fell from $724 million in 1981 to $624 million in 2011 (Table 5.2). Provincially, Ontario reported benefits that declined from $445 million in 1981 to $378 in 2011, representing a 15% reduction. Quebec also reported a decline from a benefit of $279 million in 1981 to $246 million in 2011, showing a 11.7% reduction.

TABLE 5.2. External benefits of landscape aesthetics, 1981 – 2011 (millions of 2012 dollars)

Province	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
AB	1,409	1,523	1,534	1,550	1,553	1,555	1,505	6.84
SK	1,913	1,961	1,981	1,959	1,936	1,917	1,839	-3.84
MB	561	571	569	570	560	569	538	-4.23
ON	445	416	402	414	403	397	378	-15.07
QC	279	268	253	255	252	255	246	-11.65
Region								
Western	3,883	4,054	4,084	4,079	4,050	4,041	3,882	-0.02
Central	724	685	655	669	655	652	624	-13.75

5.2.4 International comparisons

The Dutch agriculture landscape is well-know for its aesthetics. Estimates for the Netherlands outperformed those for Canada. Total external benefits from agricultural landscape aesthetics for Central and Western Canada were $73/ha compared to the Netherlands' $141.78/ha. This higher benefit reflects the picturesque rural landscape of Dutch agriculture which contributed to it being identified as a UNESCO world cultural heritage site (Jongeneel et al., 2016). The fact that the Netherlands has a high population density also factors into this estimate.

6 Non-monetized impacts

There are two remaining externalities that we have been unable to adequately monetize in a manner consistent with previous sections, but which are important, nonetheless. The first is the strength of rural communities, which is positive and the second is the impact of agriculture on wetlands, which is negative. Wetlands have been researched extensively in Canada, but there are still too many outstanding issues to be able to accurately monetize the impacts of agriculture on their degradation.[17] Estimating the impact of the agricultural sector on the strength of rural communities is another area of research that could be important future work.

6.1 Strength of rural communities

Agricultural income forms the economic bedrock of many rural communities, particularly in the Prairie provinces. Businesses in these communities are often a part of the agricultural sector or they rely on farmers as their primary customers. Not surprisingly, many researchers have found that the health of these rural communities are dependent on farm incomes, with population and economic prospects declining during farm crises and increasing during agricultural booms (Murdock, 1987; US Department of Agriculture, 2012; Windels, 2000). The role of agriculture in rural development is highly valued by some academics and policymakers, particularly within the European Union (Schmitz et al, 2010; Renting et al, 2009).

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[17] For a discussion around the contribution of wetlands to a sustainable agriculture see CAPI's report here: https://capi-icpa.ca/wp-content/uploads/2019/11/2019-10-09-CAPI-Wetlands-CAPI-Doctoral-Fellows-2017-19-group-paper_WEB.pdf

Notionally, agriculture is understood as a contributor to the health of rural communities. However, putting a specific value on this externality is fraught with difficulty. Given that some individuals choose to live in rural communities, even when better remunerated employment may be available to them in nearby urban settings, suggests that they place a substantial value on the rural lifestyle. One could, therefore, aggregate the benefits of rural communities across these residents to obtain a measure of the benefits these communities provide. On the other hand, many individuals are generally found to be more productive, healthier, and happier in urban environments (Glaeser, 2011).

6.2 Wetlands

Wetlands provide substantial ecological and economic benefits to society through the provision of ecosystem functions and services, such as wildlife habitat, water storage and filtration and carbon sequestration. Wetlands located on agricultural land can provide a range of positive externalities and yet agricultural production and the expansion of agriculture are the primary drivers of wetland loss and degradation in Canada. We are unable to provide a credible assessment of the monetary value of agriculture's net impact on wetlands due to information gaps on both the biophysical extent of agriculture's impact on wetlands as well as monetary estimates of these impacts. On the biophysical side, there is currently no comprehensive wetland inventory for Canada, nor time-series information that would allow us to quantify the change in the impacts of agriculture over time. The Canadian Wetland Inventory, a joint initiative of Ducks Unlimited Canada, ECCC, the Canadian Space Agency and the North American Wetlands Conservation Council, is a currently under development with the goal of filling this knowledge gap.[18]

In terms of wetland valuation, there have been four wetland valuation studies conducted in Canada including by Pattison et al. (2011) in Manitoba, Dias and Belcher (2015) in Saskatchewan, Lantz et al., (2013) in Ontario, and He et al.,

[18] For a discussion around the contribution of wetlands to a sustainable agriculture see CAPI's report here: https://capi-icpa.ca/wp-content/uploads/2019/11/2019-10-09-CAPI-Wetlands-CAPI-Doctoral-Fellows-2017-19-group-paper_WEB.pdf

(2017) in Quebec. All four studies use stated preference methods and estimate the public's WTP for wetland conservation programs in their respective provinces. While these studies demonstrate the economic value of Canadian wetlands to society, transferring these value estimates for our purposes is made more difficult because these WTP estimates are for specific wetland conservation programs. Some of these household values could be converted to dollars per hectare metrics to be used in a value transfer exercise. However, this assumes that people's WTP are constant over the scale of improvement, that all wetlands provide the same ecosystem services, and that the value of a wetland is independent of neighbouring population and socio-economic conditions.

The lack of information on the quantitative impact further exacerbates these value transfer issues. As a result, we did not value agriculture's impact on wetlands and identify this as a key area for future work.

7 Summary and conclusions

7.1 Summary

In the following four tables, we provide a broad overview of the externality estimates discussed above. In Table 7.1, we group all negative and positive externalities by province over time to get a sense of how externalities vary across provinces and over time.[19] The far right column lists the percentage change from the first measurement in 1981 to the measurement in 2011.[20] The table is grouped

[19] The reader will note that despite providing international comparisons in each subsection, we neglect to provide summary tables comparing Canada to other countries. This is deliberate—we feel that cross-country analysis is too opaque and inaccurate to present without debilitating caveats. Differences across countries with respect to physical externality measurement methods, land base, acres allocated to agriculture, crop selection, soil fertility, soil type, and cultural attitudes (which strongly influence stated-preference surveys) prevent accurate comparisons across countries at the level of aggregation shown in the summary tables. For more detail on international comparisons, the reader is encouraged to consult the appropriate subsections under each externality type.
[20] When the base year is negative, percentage change is not reported.

into three panes, (1) negative externalities, (2) positive externalities, and (3) net externalities—consisting of the sum of negative and positive externalities. Saskatchewan began the reporting period with the largest negative externalities in 1981. By 2011, its negative externalities were still the largest, but much more in line with Alberta and Ontario. Western Canada maintained higher levels of negative externalities throughout the reporting period. Examining net externalities reveals that agriculture in Central Canada led to higher external costs than did Western Canada, at a ratio of over 2:1; a finding that we address shortly. Aggregating both Western and Central Canadian provinces together yields a total negative net externality of approximately $4.3 billion dollars (Table 7.4).

TABLE 7.1. Negative, positive, and net environmental externalities, grouped by province over time (millions of 2012 dollars)

Province	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
				Negative externalities				
AB	2,652	2,361	2,277	2,226	2,005	2,319	2,010	-24%
SK	4,163	3,715	3,413	3,227	2,837	2,611	2,214	-47%
MB	1,021	904	835	882	910	1,054	956	-6%
ON	2,396	2,067	2,014	1,972	1,965	2,304	2,180	-9%
QC	1,219	995	997	1,059	1,183	1,479	1,496	23%
Region								
Western	7,837	6,981	6,525	6,335	5,753	5,985	5,181	-34%
Central	3,616	3,062	3,011	3,030	3,149	3,783	3,676	2%
Total negative	**11,452**	**10,043**	**9,535**	**9,365**	**8,901**	**9,768**	**8,856**	**-23%**

Province	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
				Positive externalities				
AB	1,409	1,523	1,535	1,551	1,554	1,556	1,505	7%
SK	1,916	1,964	1,984	1,962	1,939	1,920	1,842	-4%
MB	562	571	570	570	561	569	538	-4%
ON	479	448	432	445	433	427	407	-15%
QC	279	269	253	255	252	256	247	-12%
Region								
Western	3,887	4,058	4,088	4,083	4,054	4,045	3,886	0%
Central	758	716	685	700	686	683	653	-14%
Total positive	**4,644**	**4,774**	**4,773**	**4,783**	**4,739**	**4,728**	**4,539**	**-2%**

	Net externalities (positive less negative)							Percentage change
Province	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**(1981-2011)**
AB	-1,243	-838	-742	-675	-452	-764	-505	-
SK	-2,247	-1,751	-1,430	-1,266	-898	-691	-372	-
MB	-459	-333	-265	-311	-349	-485	-418	-
ON	-1,918	-1,620	-1,582	-1,526	-1,532	-1,877	-1,773	-
QC	-940	-726	-744	-804	-931	-1,224	-1,249	-
Region								
Western	-3,950	-2,923	-2,437	-2,252	-1,699	-1,940	-1,295	-
Central	-2,858	-2,346	-2,325	-2,330	-2,463	-3,100	-3,023	-
Total net	**-6,808**	**-5,269**	**-4,762**	**-4,582**	**-4,162**	**-5,040**	**-4,318**	**-**

In Table 7.2, Table 7.3, and Table 7.4 each externality is analyzed over time for the Western provinces, the Central provinces, and for Canada (the sum of Western and Central Canada estimates). Table 7.2 examines each externality estimate in Western Canada, revealing the large contribution of PM emissions and soil erosion to the estimate of total negative externalities. For example, in 1981, the external costs of PM emissions were valued at C$4 billion, almost double the valuation of the next largest category (i.e. soil erosion at $2.2 billion). By 2011, the damage from both soil erosion and PM emissions decreased significantly, but they still constituted the largest factors influencing the value of negative externalities for the Western provinces. Positive externalities for these provinces are dominated by landscape aesthetics, the most controversial and uncertain estimate. If beauty is in the eye of the beholder, it is clear that the value of landscape aesthetics will vary dramatically from person to person, and as discussed in Section 5.2.2, the most appropriate valuation study available may still not capture the true value of landscape aesthetics. As it stands, our estimates suggest that the positive value of landscape aesthetics outweighs the damages from both PM emissions *and* soil erosion, which certainly deserves further attention in subsequent research.

TABLE 7.2. Negative, positive, and net environmental externalities over time for the Western Provinces: Alberta, Saskatchewan, and Manitoba (millions of 2012 dollars)

	Negative externalities							
Externality	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**Percentage change (1981-2011)**
GHG[21]	869	845	869	1,001	896	837	732	-16%
Ammonia	670					1,087	962	44%
PM	3,783	3,451	3,084	2,810	2,377	1,907	1,462	-61%
N-water	43	69	44	82	132	153	188	336%
P-water	39	43	44	46	48	47	46	17%
Pest-water	118	130	137	146	158	174	188	59%
Coliform-water	38	37	38	39	39	41	40	5%
Soil erosion	2,230	2,357	2,260	2,162	2,054	1,689	1,514	-32%
Wildlife/biodiversity	47	49	49	49	49	49	47	1%
Total negative	**7,837**	**6,981**	**6,525**	**6,335**	**5,753**	**5,985**	**5,181**	**-34%**

	Positive externalities							
Externality	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**Percentage change (1981-2011)**
Wildlife habitat	3.75	3.86	3.90	3.87	3.82	3.80	3.64	-3%
Landscape aesthetics	3,883	4,054	4,084	4,079	4,050	4,041	3,882	0%
Total positive	**3,887**	**4,058**	**4,088**	**4,083**	**4,054**	**4,045**	**3,886**	**0%**
Total net	**-3,950**	**-2,923**	**-2,437**	**-2,252**	**-1,699**	**-1,940**	**-1,295**	**-**

In Table 7.3, we examine each externality over time for Central Canada, Ontario and Quebec. For these provinces, the biggest negative externality is GHG emissions, which is commensurate with these provinces' higher use of N fertilizer and related N_2O emissions. The positive effects of landscape aesthetics provide some balance to offset these estimates, but not nearly as much as for the Western provinces, which is largely a result of the large land base devoted to agriculture and not a reflection of a more or less "pleasing" aesthetic across regions. The net

[21] The values shown in Table 7.2, Table 7.3, and Table 7.4 are the cross-Canadian averages with different parts of the country having vastly different externality values due to farming methods, soil types, and other factors like farming intensity and climate.

externalities for Central Canada suggest higher impacts from agriculture compared to the Western provinces. But again, we stress that this result is being driven by the high valuation of landscape aesthetics in Western Canada.

TABLE 7.3. Negative, positive, and net environmental externalities over time for the Central Provinces: Ontario and Quebec (millions of 2012 dollars)

Externality	1981	1986	1991	1996	2001	2006	2011	Percentage change (1981-2011)
				Negative externalities				
GHG	810	783	740	767	763	791	771	-5%
Ammonia	650					608	537	-17%
PM	206	199	194	176	167	154	139	-32%
N-water	663	788	762	728	810	828	796	20%
P-water	9	8	8	8	9	9	9	2%
Pest-water	421	462	518	555	596	639	681	62%
Coliform-water	5	4	4	3	3	2	2	-58%
Soil erosion	613	593	568	571	583	537	535	-13%
Wildlife/biodiversity	239	225	216	222	217	215	205	-14%
Total negative	**3,616**	**3,062**	**3,011**	**3,030**	**3,149**	**3,783**	**3,676**	**2%**
				Positive externalities				
Externality	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**Percentage change (1981-2011)**
Wildlife habitat	34	32	31	32	31	30	29	-15%
Landscape aesthetics	724	685	655	669	655	652	624	-14%
Total positive	**758**	**716**	**685**	**700**	**686**	**683**	**653**	**-14%**
Total net	**-2,858**	**-2,346**	**-2,325**	**-2,330**	**-2,463**	**-3,100**	**-3,023**	**6%**

In Table 7.4, we add both regions together to examine the value of each externality for the most agriculturally-intensive provinces in Canada. At this scale, we see a slightly different picture of the relative impacts from different categories of externalities. Aggregating all five provinces together, we see that soil erosion has the most damaging impacts, with a value of over C$2 billion in 2012 dollars. This is followed by the negative impacts of PM pollution, GHG emissions and NH_3 emissions. However, if we add up all of the separate water-related damages

(N, P, pesticide, and coliform) we see an aggregate estimate related to water damage from agriculture of C$1.9 billion, surpassing GHG emissions impacts and just trailing soil erosion (which also contributes to water pollution). In sum, total negative externalities add up to almost C$9 billion in 2012 dollars, which is offset by about C$4.5 billion in positive externalities for a total net external cost of just over C$4 billion dollars in 2011.

Closer inspection of Table 7.4 yields some positive trends. With the exception of the external costs associated with NH_3, N, P, and coliform water pollution, negative externalities have decreased across Canada from 1981 to 2011. Cumulatively, the damages from negative externalities have declined by about 23%, representing significant effort by the sector to address its negative environmental impacts. The most notable improvements are from the impacts of PM emissions and soil erosion, which declined by 60% and 28% respectively. Much of this can be attributed to the adoption of zero-till practices in the Prairie provinces which led to reductions in both PM emissions and soil erosion.

TABLE 7.4. Negative, positive, and net environmental externalities over time for the Western and Central Provinces (millions of 2012 dollars)

	Negative externalities							
Externality	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**Percentage change (1981-2011)**
GHG	1,679	1,628	1,609	1,768	1,659	1,628	1,503	-10%
Ammonia	1,319					1,696	1,499	14%
PM	3,989	3,651	3,278	2,986	2,544	2,061	1,601	-60%
N-water	706	857	806	810	942	981	985	39%
P-water	48	52	52	54	57	56	55	14%
Pest-water	539	592	655	701	754	813	869	61%
Coliform-water	43	41	42	42	42	44	42	-3%
Soil erosion	2,843	2,950	2,828	2,733	2,637	2,226	2,049	-28%
Wildlife/biodiversity	286	274	266	271	266	264	253	-12%
Total negative	**11,452**	**10,043**	**9,535**	**9,365**	**8,901**	**9,768**	**8,856**	**-23%**
	Positive externalities							
Externality	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**Percentage change (1981-2011)**
Wildlife habitat	38	36	35	35	35	34	32	-6%

Landscape aesthetics	4,607	4,739	4,739	4,748	4,705	4,693	4,506	-4%
Total positive	**4,644**	**4,774**	**4,773**	**4,783**	**4,739**	**4,728**	**4,539**	**-2%**
Total net	**-6,808**	**-5,269**	**-4,762**	**-4,582**	**-4,162**	**-5,040**	**-4,318**	**-37%**

7.2 Conclusions

Table 7.4 therefore represents a summary of the findings from this report can also be viewed as guidance for a set of policy priorities for addressing agricultural externalities in Canada since it shows the relative magnitude of agriculture's environmental impacts. What is missing from the table is however, the cost of mitigation. In order for policies that address these externalities to be efficient, they must reduce these impacts at the lowest possible unit cost. While soil erosion appears to impose the highest external costs in Canada, this does not imply that it should be the first policy priority. The next logical step is to calculate the marginal abatement cost for each of these externalities (i.e. the cost to society of reducing one unit of externality). With this information, we could rank each externality by the ratio of marginal cost to society to marginal cost of mitigation or abatement. This would provide us with a place to start in terms of identifying policy priorities for addressing the externalities from agriculture. In this report, we have calculated the numerator of this all-important ratio (i.e. cost to society). In future research, we need to address the denominator (i.e. cost of mitigation).

Another interesting policy implication from these results is the extent to which the values of externalities vary across regions and provinces. Apart from the impacts of agriculture on water quality, which are an issue in all provinces, the impacts from soil erosion are the largest external cost in the Western provinces while the impact of GHG emissions from agriculture are the largest cost in Central Canada. This heterogeneity suggests that provincial-level policy approaches will most likely differ from federal-level policy approaches to address the issues most relevant for each region. Further, with the exception of GHG emissions, each of the externalities we address in this report has localized effects, which further suggests provincial-specific targeting. The fact that GHG emissions impose external costs regardless of their origin makes it the most difficult externality to address—Western provinces will have little appetite to reduce GHG emissions when all of their work could be undone by inaction in Central Canada (or vice versa). This suggests that a national approach to GHG emissions may be the most efficient policy solution.

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9 Appendix

9.1 Methods for GIS analysis of Crop Type and GHG emissions on Agricultural Land across Canada

Geo-spatial analysis was conducted to investigate the links between GHG emissions and crop type. The geo-spatial analysis was conducted using ArcMap 10.5. Three key datasets were used for this analysis:

- Provincial boundaries (Statistics Canada 2011);
- The GHG emissions layer of Agriculture and Agri-Food Canada's Agri-Environmental Indicators dataset (Agriculture and Agri-Food Canada, 2016); and
- The 2011 Crop Inventory (Agriculture and Agri-Food Canada, 2011).

All maps were analyzed using the Albers Conic Equal Area Projected Coordinate System and the WGS 1984 Geographic Coordinate System. All area calculations were conducted using the projected coordinate system. Analyses were conducted separately for each province. Crop inventory was not available in 2011 for Newfoundland, Nunavut, the North West Territories, or the Yukon. The only year with overlapping information for both datasets was 2011, which is why that year was chosen for analysis.

To conduct the analysis, the crop inventory data layer was converted from a raster dataset to a vector dataset. Provincial boundaries were used to clip the Canada wide GHG layer into separate GHG layers for each province. The intersect function was used to create a map layer with new polygons containing both crop type information and GHG emissions information. The geometry of these new polygons was calculated, and other characteristics were measured using the field calculator.

Both average GHG emissions and the distribution of GHG emissions on land covered by a particular crop were investigated. The GHG dataset contained the GHG emissions rate (in kilograms of CO_2 equivalent per hectare, or KgC02eq/ha) for each map polygon. It also categorized each polygon with a code (from 1-5) indicating the relative intensity of emissions from that polygon (very low to very high). An estimate of the KgC02eq for each polygon was estimated by multiplying the KgEqC02/ha of its 'parent' polygon by the size of the new polygon. In the final set of geo-processing, the dissolve tool was used to combine all polygons that had both the same crop type and the same emission intensity code. The total number of $KgEqC0_2$ for a particular "crop type + GHG emission intensity" combination was also added up during this process. The resulting table was then exported to excel for further analysis.

The crop inventory dataset included information on both agricultural and non-agricultural land uses. There were different levels of agricultural land analysis, depending on the types of crops grown in the province, as described in Section 4.2.1 of the Annual Crop Inventory Data Product Specifications (Agriculture and Agri-Food Canada 2011b). For example, most of the Eastern provinces have an estimate for the total cover of 'cereal' grains, while other provinces have estimates for the specific grain types within that classification (barley, millet, oats, rye, spelt, triticale, wheat, switchgrass, sorghum, quinoa, winter wheat, sprint wheat, and other grains).

For the purposes of this analysis, the distinction between 'agricultural' and 'other' land uses was made based on whether the land is, or is likely in the near future, to be used for agricultural purposes. This includes all crops. In addition, 'Too Wet to be Seeded' and 'Fallow' land would not produce crops in 2011, but they are likely to be seeded in the future, so they are included in the assessment of agricultural land. However, 'exposed land/barren' is not likely to produce crops in the near future and was assessed as being an 'other land use'. Grassland refers to predominantly native grasses and was not assessed as being agricultural. Pasture/forages refers to periodically cultivated land that is grazed and was therefore assessed as being agricultural land.

The distribution of land in a particular crop across the five different GHG categories was assessed by dividing the area of land with a particular land use and emission category by the total amount of land with that land use. These distributions are shown in bar graphs for each province.

In addition, the average GHG emission intensity (kgC0_2eq/ha) for each land use category was calculated. This value is then compared to the average emission intensity for grassland within that province. Grassland is seen as being a proxy for land that is undisturbed by agriculture, and which is more comparable to crop emissions than forested land would be.

9.2 Quality Checking

Several methods were used to assess the accuracy of this analysis:

- The total area of land before and after geo-spatial processing was measured and were no more than 0.1% different from each other.
- Similarly, the total estimated KgC0_2eq per province were measured before and after geo-spatial processing and were <0.01% different from each other.
- The estimated accuracy of the original 2011 crop inventory layer was referenced. The average estimated accuracy of the base crop layer in 2011 was 80%, with a range from 67%-88%. The accuracy of crop estimates was not evaluated in BC in 2011.
- For reference, the seeded area of two to three crops in the 2010/2011 crop year are reported for each province (Statistics Canada 2019). While these values are almost always within the same order of magnitude as the area sizes in the GIS layers, there is still sometimes a difference of 50% or more between these values. This is likely due to different characterizations of some land – for example, some crop inventory land is lumped into 'agriculture- undifferentiated' or 'other crops'. As well, the crop inventory layer itself is known to have some errors.
- The measured area of the selected crops are reported both a) from the original dataset and b) from the final, geo-processed dataset. The average percentage difference between these datasets is 7%, with values ranging from 0% (no difference) to 42% (for cereals in Quebec).

Overall, the quality check indicates that the analysis results are reasonably robust, if not 100% accurate in all cases.