Valuing Changes in Agri-Environmental Indicators

Full Cost Accounting for Agriculture – Year 2 Report

A paper written for

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Executive Summary

This paper reports on the second year of a five-year project being undertaken by the International Institute for Sustainable Development (IISD) for Agriculture and Agri-Food Canada (AAFC). The five-year project is exploring the topic of calculating the environmental externalities that arise from agriculture in Canada.

This report takes as its starting point the overview of valuation issues and methodologies done last year in 2004. The objective of this year's work was to develop a conceptual methodology for valuing changes in agri-environmental indicators and to use this methodology to identify important data needs and gaps. This year, two major tasks were undertaken toward this objective:

- The development of a conceptual methodology for valuing changes in agri-environmental indicators;
- Testing the methodology by using it to analyse five of the agri-environmental indicators, and reporting on the results.

This paper reports on the above tasks, and uses the results as the basis for suggesting focus areas for next year's work plan.

Developing the Methodology – the *Impact Pathway Approach*

The *Impact Pathway* approach was used to develop conceptual models for valuing changes in five agri-environmental indicators and for assessing the availability of required economic and physical data. This approach, and close variants of it, is frequently used in valuation studies. For purposes of this study we developed the following terminology for the various stages of the *Impact Pathway* approach:

- ***Pathway Constituent Model*** – identification of the constituents (e.g., pollutants) associated with an agri-environmental indicator which can result in an impact on ecosystems, and the routes or pathways by which the impacts can potentially occur (e.g., sediment loading for the *Risk of Water Erosion* indicator);
- ***Transport Model*** – the methodology for how the pathway constituent disperses to the location of impact in an ecosystem (e.g., converting soil loss on farms to sediment loading rates to waterways and to turbidity concentrations in waterways);
- ***Impact Model*** – the methodology for relating the concentration of the pathway constituent at the location of impact, to specific environmental and human wellbeing impacts (e.g., the impact of increased turbidity on recreational water use).
- ***Valuation Model*** – the methodology for determining the cost associated with the specific environmental or human wellbeing impacts (e.g., the willingness to pay for recreational water use).

Identifying these pathway constituents is not always an intuitive process – some will be obvious, but others will not. To facilitate the identification of impact pathways we used an ecosystem services and human wellbeing framework which has been adapted from the literature to give a comprehensive list of how environmental changes might affect people.

The details of the methodology are discussed in Section 2 of this paper.

Applying the Impact Pathway Approach

We analyzed five different agri-environmental indicators using the *Impact Pathway* approach, namely:

- Risk of Water Erosion – Soil Quality category
- Risk of Water Contamination by Phosphorous – Water Quality category
- GHG Emissions – Agro-Ecosystem Atmospheric Emissions category
- Energy Use Efficiency – Eco-Efficiency category
- Wildlife Habitat – Agricultural Biodiversity category

Each of these has different characteristics. The uniqueness of the indicators selected for analysis was evident from the impact pathway analysis. Each had unique pathway constituents leading to impacts on ecosystem services and human wellbeing, and consequently, each required its own analysis and review of available data. The key general lessons from the analysis were that the methodology was robust, in that it guided the work on a disparate set of indicators and was helpful in identifying data needs and gaps. Furthermore, the analysis revealed that defining the full set of impact pathways for each indicator is a complex and time consuming job. In part, the job is complex because there is a good deal of prior research that can help answer the valuation questions. However, it must be organized and applied to specific situations, and the remaining gaps identified.

Key Results

A sampling of key points emerging from the impact pathway analyses, as articulated in Section 9, are provided below.

- ***Pathway Constituent Modelling***
  The use of an ecosystem services and human wellbeing framework proved robust for helping to identify detailed impact pathways. While many of the impact pathways were intuitive and could have been identified without using an ecosystem and human wellbeing framework, the framework was helpful in the brainstorming process. The ecosystem and human wellbeing framework we used operates at a fairly high level of generality, and contains some redundancy, but serves the purpose of ensuring that important impacts are not overlooked.

- ***Transport Modelling***
  Of the five agri-environmental indicators studied, the output from three of the indicators can be used directly in impact and valuation modelling. These include

the *Availability of Wildlife Habitat*, *GHG Emissions*, and *Energy Efficiency* indicators. The output from the *Risk of Water Erosion* and the *Risk of Water Contamination by Phosphorus* indicators will require additional transport modelling in order to assess the change in the state of the ecosystem necessary for determining impacts to ecosystem goods and services. Empirical and physically-based methods have been developed for such modelling, and it will be a matter of determining which methods are feasible given the need to replicate the analysis and aggregate to a national scale.

- ***Impact and Valuation Modelling***
  The methods for impact and valuation modelling are relatively well established for most of the impact pathways analysed in this report. Data are also available from studies conducted in Canada, the United States and internationally to allow a first level valuation. Benefit transfer appears quite common in Canadian valuation exercises, particularly related to water erosion and habitat changes. Many of the valuation exercises cited in the literature can be traced back to a relatively small subset of primary data sources. It is also the case that many of the assumptions upon which the primary data sources are not carried forward in the benefit transfer process, making it difficult to assess the credibility of the valuation.

  The watershed or water basin spatial units can potentially serve as an important unit of analysis for valuing changes in agri-environmental indicators. This unit of analysis was used in the literature for valuing changes in two of the five indicators studied – namely the water related indicators (e.g., the risk of water erosion and the risk of water contamination by phosphorus indicators). Given that such a unit of analysis would likely be required for all indicators related to soil and water quality, it may prove convenient to do so for the other indicators, but this remains to be tested.

- ***Linkages to Integrated Economic-Environmental Modelling***
  For macro-level integrated modelling, the Land Use Allocation Model (LUAM) appears to be a promising channel whereby policy changes simulated via the Canadian Regional Agricultural Model (CRAM) model can communicate with changes in agri-environmental indicators at the SLC polygon or watershed/basin level.

- ***Linkages to Social Indicators***
  The use of an ecosystem services framework for identifying impact pathways can be a useful mechanism for identifying important linkages and feedback loops between changes in agri-environmental indicators and AAFC's social indicators relating to human wellbeing.

Results of the impact pathway analysis for each of the five agri-environmental indicators are presented in Sections 3 through 7. Key findings include the following:

- ***Risk of Water Erosion***
  - Valuing changes in this indicator will require development of a transport model component to convert soil loss from farmland to changes in sediment and turbidity in channels and major waterways.
  - The pathway constituents for this indicator include changes in phosphorus and nitrogen loading to water, in addition to sediment loading and soil productivity loss. Therefore, valuing changes in the *water erosion* indicator need to incorporate the results from the valuations in the *risk of water contamination by phosphorus and nitrogen indicators*, and potentially the new water quality indicators being developed.

- ***Risk of Water Contamination by Phosphorus***
  - The existing indicator measures a risk of pollution, and does not contain an evaluation of actual phosphorous loading to water, which is necessary for an estimate of ecosystem impact and the associated costs and benefits. Research is underway to make the links, but it is not yet complete.
  - A range of methods are and can be considered to close this gap – among them are relating the risk indicator directly to water quality in waterways, or determining loading rates based on the indicator and transporting this load to water ways using empirically or physically-based models.

- ***Availability of Wildlife Habitat on Farmland***
  - The most influential farmland habitat types impacting on wildlife are wetlands and woodlots. The impact pathways associated with changes in these habitat areas are numerous and there is appreciable amount of valuation data available that could be used for benefit transfer. However, changes in these habitat areas cannot be determined using the current indicator (a factor of the availability of data in the Census of Agriculture). This data gap will have to be overcome before realistic valuation estimates can be made for this indicator.

- ***GHG Emissions***
  - While there are no methodological or data gaps that would prevent valuation of changes in the GHG emissions indicator, the issue of human mortality valuation on a global scale is problematic. The Intergovernmental Panel on Climate Change is currently avoiding a comprehensive global valuation of climate change damages for this reason.
  - A non-rigorous, but much less politically problematic approach may well be to simply value agricultural GHG emissions at the international market price for emissions credits.

- ***Energy Efficiency***
  - We examined the impacts of agriculture on ecosystems caused by the impacts of the energy used in agriculture, at all stages before the energy arrives at the farm. This is the only indicator studied that dealt with

inputs. A life cycle approach implies that a great many impact pathways are involved and ultimately costs must be developed for all of the important ones. This is a complex and time-consuming analysis.

- However, a great deal of work has already been done on many of the impact pathways that will need to be evaluated, so while there would doubtless be important data gaps, policy relevant costing information can likely be found.

Possible Next Steps

The analyses conducted this year has helped advance our thinking related to the *big picture* of valuing changes in agri-environmental indicators. The process for valuing changes in agri-environmental indicators can be thought of as consisting of various levels of analysis as illustrated on Figure ES-1. This paper reports on the indicator level analysis which was designed to explore the conceptual methods and data needs and gaps associated with the valuation of indicator changes.

A next level of analysis could use the *Impact Pathway* approach to actually calculate the total value of a change in one selected indicator – with reference to a specific location, such as a specific watershed basin. As illustrated on Figure ES-1, this would entail an iterative process whereby the specific methods selected for the transport, impact and valuation modelling would be determined keeping in mind the feasibility of replicating the calculation for all watershed basins in a province. Additionally, the ability to provide relevant feedback to macro-level integrated economic-environmental modelling (e.g., via CRAM and LUAM) would be explored and taken into consideration in the selection of specific methods and calculation of the total value for changes in the selected indicator.

This type of basin-level analysis could potentially be a focus for our next year's work plan. The result from this work would move the process one step closer to carrying out a provincial-level valuation calculation, and then toward a full national-level valuation of the selected indicator. The methods developed and lessons learned through these studies would inform similar analyses to be conducted for the remaining agri-environmental indicators.

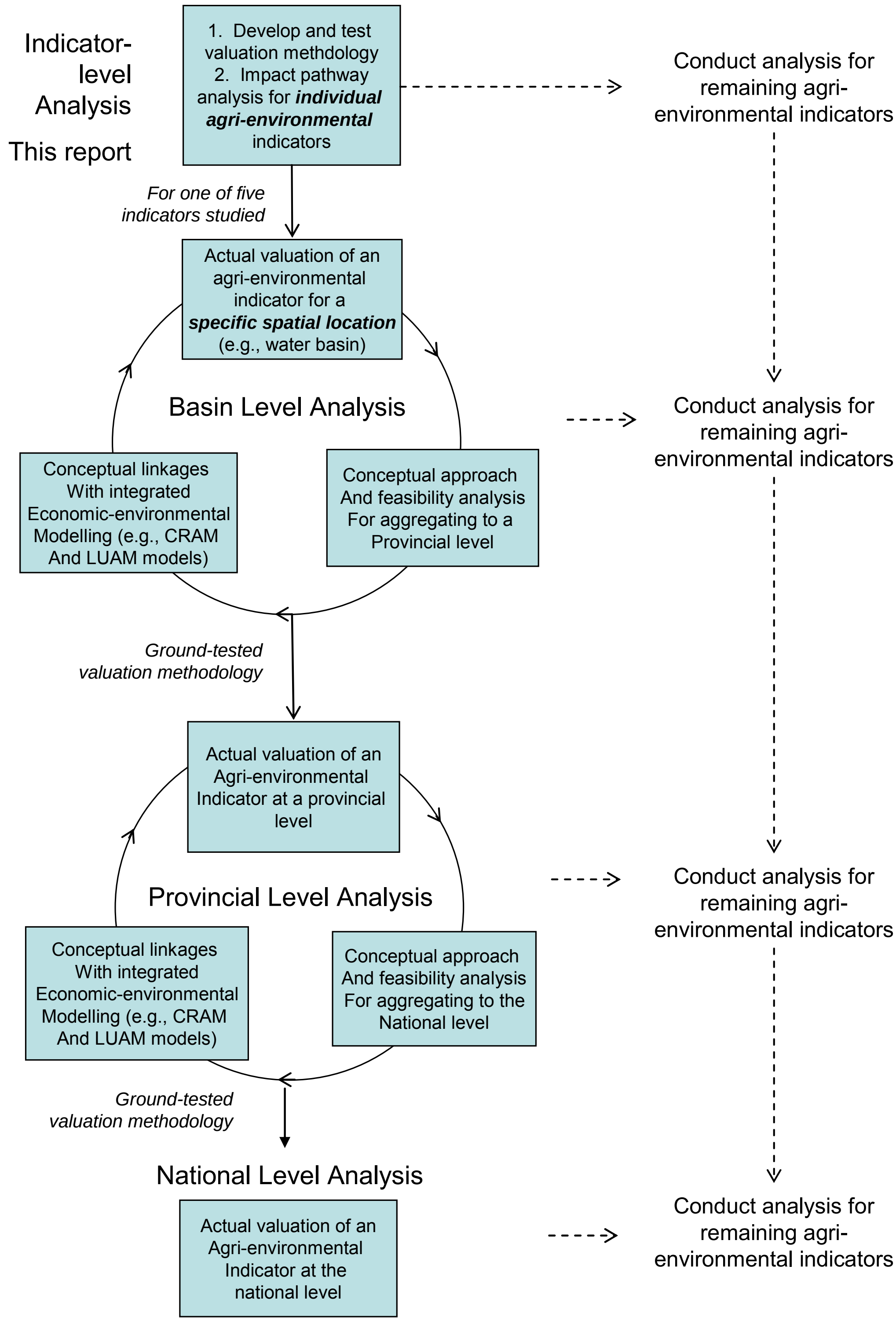

Figure ES-1. IISD's conceptualization of a process for advancing towards the valuation of changes in agri-environmental indicators at the national level.

1. Introduction

This paper is second in a series of papers on the full-cost accounting of agriculture prepared under the joint AAFC-IISD multi-year work agreement. The first paper, produced in year 1 of the work agreement, provided a review of the pertinent literature on the topic of economic valuation of the off-farm environmental costs and benefits associated with agriculture operations (IISD 2004). Full-cost accounting work in year 2 focused on developing a methodology for, and assessing the feasibility of, valuing changes in a select number of agri-environmental indicators developed under the NAHARP program.[1] For example, if the *Risk of Water Erosion* indicator were to show a certain percentage decrease, what would be the economic benefits of doing so – including both on-farm costs and benefits (i.e., internal private costs) and off-farm costs and benefits (external social costs).

The specific objectives of this year 2 work include the following:

- To develop preliminary conceptual models for valuing the economic benefits and costs associated with changes in agri-environmental indicators; and
- To review the available economic and physical data required for such valuation.

Five indicators were selected for detailed conceptual analysis, one for each major agri-environmental indicator category. These five focus indicators were:

- Risk of Water Erosion – Soil Quality category
- Risk of Water Contamination by Phosphorous – Water Quality category
- GHG Emissions – Agro-Ecosystem Atmospheric Emissions category
- Energy Use Efficiency – Eco-Efficiency category
- Wildlife Habitat – Agricultural Biodiversity category

Full-cost accounting or economic valuation of environmental impacts is important for many reasons. For example:

- ***Highlights issues*** - Many environmental problems have complex causes, and thus we are sometimes surprised when they arise. One advantage of a full-cost accounting is that it forces us to look at issues comprehensively.
- ***Informs policy makers re priorities*** – Information from full-cost accounting can act as a very useful guide in policy making, in that it can provide a fairly objective basis for setting priorities. It is often the case that a problem may have a high media or political profile, and thus get the most policy attention.
- ***Improves public discussion*** - The existence of specific data on the costs and benefits of various courses of action will help improve public discussion about the

[1] The National Agri-Environmental Health Analysis and Reporting Program (NAHARP) was established to help *strengthen departmental capacity in the development and continuous improvement of agri-environmental indicators and of the tools that use these indicators in policy development and integrated the environment and the economy*.

options. Sometimes the public debate is based on a series of hypothetical statements about costs and benefits, over which the proponents of various viewpoints can argue but not agree. If there are useful numbers to attach to the discussion, then it can focus more on issues and less on which hypothetical statement is most accurate. Of course, this depends on having a set of reasonably agreed numbers.

- ***Informs policy design*** - The analysis will give information on both the sources of problems and on those who bear the burden of the costs. This can be very helpful in designing policies that might alleviate the problem, for two reasons. First, the policy responses can be aimed at the most relevant parties; and second, the information on the amounts of costs or benefits being created can guide the type and rigour of the policy design.

The broad conceptual framework guiding the full-cost accounting work in year 2 places the decision maker at the fore – as the user of agri-environmental indicator valuation information (Figure 1-1). A decision maker could conceivably obtain this information in two ways. One is by simply knowing the on-farm and off-farm environmental costs and benefits associated with desired changes in any of the NAHARP agri-environmental indicators. This would require an analytic model for each agri-environmental indicator that couples the output of the indictor with the necessary economic and physical data to calculate the on-farm and off-farm costs and benefits.

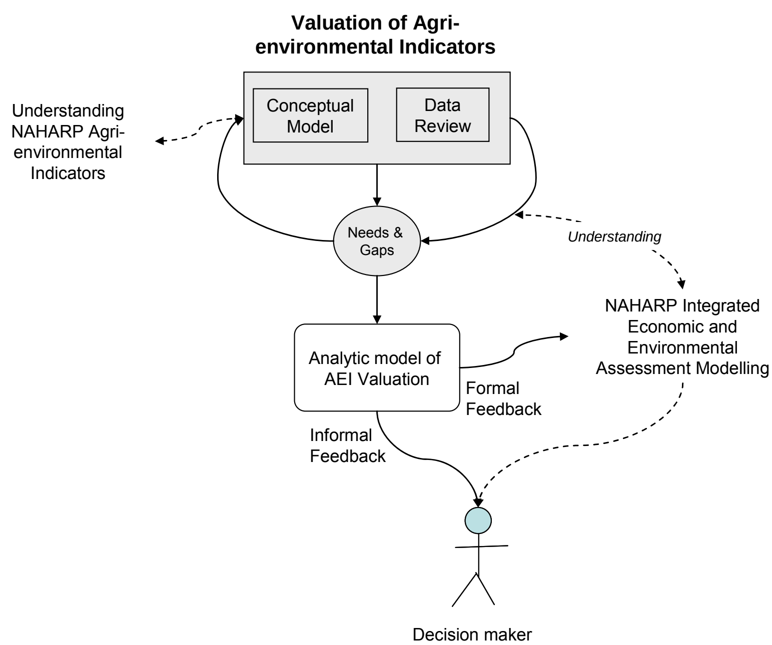

Figure 1-1. Illustration of work required for valuation of agri-environmental indicators (2004-05 work shaded in grey)

The other is via results from integrated economic and environmental modelling and forecasting, for which the full-cost accounting information could be a formal input. Current integrated modelling at AAFC employs the Canadian Regional Agricultural

Model (CRAM) as a means for predicting future effects of policies that are adopted, coupled with other models such as the Land Use Allocation Model (LUAM) to help project where specific changes might take place. Advancing tools and capacity for this type of integrated modelling is one of the three main NAHARP program activities. In this paper we provide some commentary on the potential linkages between the individual agri-environmental indicator valuation information and the integrated modelling components.

The focus of this paper is on developing conceptual models for the valuation of individual agri-environmental indicators and reviewing the availability of economic and physical data needed for the valuation. This work required developing an understanding and appreciation of the methods and future directions for: 1) calculating the agri-environmental indicators; and 2) the needs of the integrated economic and environmental modeling component of the NAHARP program. In this paper we identify the main methodological and data gaps that currently exist for valuing changes in the five focus indicators. We conclude the paper with a look toward the year 3 work plan of the AAFC-IISD work agreement and with recommendations for how AAFC might proceed with developing analytic models for valuing changes in all of the agri-environmental indicators.

2. Methods of Analysis – the Impact Pathway Approach

The *Impact Pathway* approach is used to develop conceptual models for valuing changes in the five focus agri-environmental indicators and for assessing the availability of required economic and physical data.

In determining the off-farm costs and benefits of an activity (i.e., externalities), one must define how the activity affects the environment and human wellbeing. This requires defining the impact pathways – the routes by which the actual damage and benefit takes place. For example, recent work in both the United States and the European Union on calculating the externalities of electricity production developed the conceptual diagram shown on Figure 2-1. In this example, emissions of SO_2 represent one pathway by which the activity of electricity production can impact on the environment. The SO_2 emissions dispersed through the air to a specific location whereby a specific impact on the environment or human health occurs. This impact could be a change in crop yield or it could be an increase in the occurrence of asthma in nearby cities due to an increase in SO_2 concentrations. Finally, the valuation exercise is completed by knowing the cost associated with the impact – for example, by multiplying the change in the number of asthma cases by the willingness-to-pay (WTP) to avoid those cases.

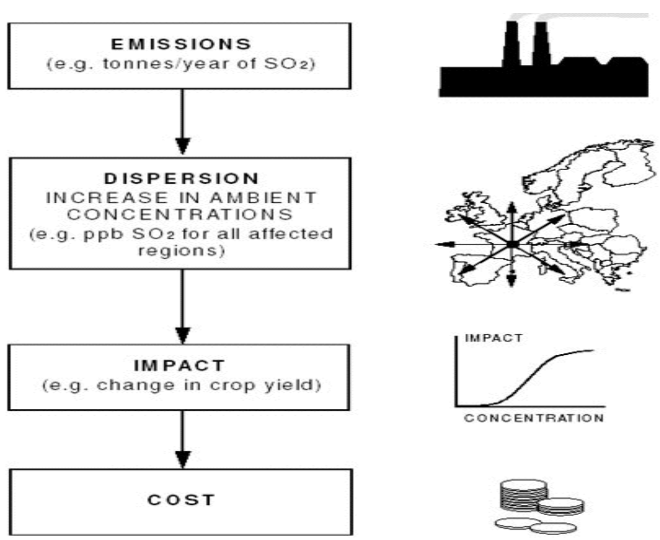

Figure 2-1. Impact-pathway methodology (Holland et al. 1999).

For purposes of our study we have developed the following terminology for the various stages of the Impact Pathway approach.

- ***Pathway Constituent Model*** – identification of the constituents and routes by which an impact can occur.

- ***Transport Model*** – the methodology for how the pathway constituent disperses to the location of impact environmental or human wellbeing.
- ***Impact Model*** – the methodology for relating the level of the pathway constituent at the location of impact to the specific environmental and human wellbeing impacts.
- ***Valuation Model*** – the methodology for determining the cost associated with the specific environmental or human wellbeing impact.

2.1 Pathway Constituent Model

The Pathway Constituent Model component of the approach is designed to identify the modes by which impact resulting from a change in the agri-environmental indicator can occur. For example, for SO_2 emissions would be a pathway constituent for a change in the amount of electricity production. Other pathway constituents for electricity production could be GHG emissions and water quality (e.g., changes in temperature from coolant water discharge to rivers).

Identifying these pathway constituents is not always an intuitive process – some will be obvious, but others will not. To facilitate the identification of pathways we used an ecosystem services and human wellbeing framework. This is helpful in the brainstorming process because it looks at the pathway from the perspective of a list of potential impacts on the environment and human wellbeing. The framework we used is based on the general framework used in the Millennium Ecosystem Assessment (MA 2003), shown on Figure 2-2. This framework allows the identification of impacts to the range of ecosystem services, which in turn have impact on various aspects of human wellbeing. Using this approach, we developed a table to facilitate a brainstorming session for identifying potential impact pathways. This table is presented in Table 2-1. To define the specific ecosystem services we use the terminology of de Groot et al. (2002). Annotative descriptions of the various ecosystem goods and services are provided in Table 2-2. The human wellbeing framework is adopted from the Millennium Ecosystem Assessment (MA 2003).

2.2 Transport Model

The transport model component establishes a methodology for how the pathway constituent travels to the location of impact. This component will not be required for instances where the agri-environmental indicator describes the change in the state of the ecosystem which is directly linked to the impact on an ecosystem service of aspect of human wellbeing. For example, in the case of SO_2 emissions, if the indicator of interest described changes in urban SO_2 concentrations, a transport model would not be required. However, if the indicator of interest describes SO_2 emissions from a plant, the only way to arrive at off-site external costs is to determine how the SO_2 emissions are linked to changes in urban SO_2 concentrations.

Table 2-1. Framework for facilitating the identification of impact pathways

	Potential Impacts
Ecosystem Goods & Services	Indicator Name
Regulation Functions	
1. Gas regulation	
2. Climate regulation	
3. Disturbance prevention	
4. Water regulation	
5. Water supply	
6. Soil retention	
7. Soil formation	
8. Nutrient regulation	
9. Waste treatment	
10. Pollination	
11. Biological control	
Habitat Functions	
12. Refugium function	
13. Nursery function	
Production Functions	
14. Food	
15. Raw materials	
16. Genetic resources	
17. Medicinal resources	
18. Ornamental resources	
Information Functions	
19. Aesthetic information	
20. Recreation	
21. Cultural and artistic information	
22. Spiritual and historic information	
23. Science and education	
Human Wellbeing	
Security	
Ability to live in an environmentally clean and safe shelter	
Ability to reduce vulnerability to ecological shocks and stress	
Basice material for a good life	
Ability to access resources to earn income and gain livelihood	
Health	
Ability to be adequately nourished	
Ability to be free from avoidable diseases	
Ability to have adequate and clean drinking water	
Ability to have clean air	
Ability to have energy to keep warm and cool	
Good social relations	
Opportunity to express aesthetic and recreational values associated with ecosystems	
Opportunity to express cultural and spiritual values associed with ecosystems	
Opportunity to observe, study and learn about ecosystems	

Note: The ecosystem services framework is based on valuation work reported by de Groot et al. (2002). The human wellbeing framework is adopted from the Millennium Ecosystem Assessment (MA 2003).

Table 2-2. Ecosystem functions, goods and services of natural and semi-natural ecosystems (from de Groot et al. 2002)

Functions	Ecosystems processes and components	Goods and Services (examples)
Regulation Functions	***Maintenance of essential ecological processes and life support systems***	
1. Gas regulation	Role of ecosystems in bio-geochemical cycles (e.g. CO_2/O_2 balance, ozone layer, etc.)	1.1 UVb-protection by O_3 (preventing disease). 1.2 Maintenance of (good) air quality. 1.3 Influence on climate (see also function 2.)
2. Climate regulation	Influence of land cover and boil. Mediated processes (e.g. DMS-production) on climate	Maintenance of a favourable climate (temp., precipitation, etc) for, for example, human habitation, health, cultivation
3. Disturbance prevention	Influence of ecosystem structure on dampening env. disturbances	3.1 Storm protection (e.g. by coral reefs). 3.2 Flood prevention (e.g. by wetlands and forests)
4. Water regulation	Role of land cover in regulating runoff & river discharge	4.1 Drainage and natural irrigation. 4.2 Medium for transport
5. Water supply	Filtering, retention and storage of fresh water (e.g. in aquifers)	Provision of water for consumptive use (e.g. drinking, irrigation and industrial use)
6. Soil retention	Role of vegetation root matrix and soil biota in soil retention	6.1 Maintenance of arable land. 6.2 Prevention of damage from erosion/siltation
7. Soil formation	Weathering of rock, accumulation of organic matter	7.1 Maintenance of productivity on arable land. 7.2 Maintenance of natural productive soils
8. Nutrient regulation	Role of biota in storage and re-cycling of nutrients (eg. N,P&S)	Maintenance of healthy soils and productive ecosystems
9. Waste treatment	Role of vegetation & biota in removal or breakdown of xenic nutrients and compounds	9.1 Pollution control/detoxification. 9.2 Filtering of dust particles. 9.3 Abatement of noise pollution
10. Pollination	Role of biota in movement of floral gametes	10.1 Pollination of wild plant species. 10.2 Pollination of crops
11. Biological control	Population control through trophic-dynamic	11.1 Control of pests and diseases. 11.2 Reduction of herbivory (crop damage)
Habitat Functions	***Providing habitat (suitable living space) for wild plant and animal species***	***Maintenance of biological & genetic diversity (and thus the basis for most other functions)***
12. Refugium function	Suitable living space for wild plants and animals	Maintenance of commercially harvested species
13. Nursery function	Suitable reproduction habitat	13.1 Hunting, gathering of fish, game, fruits, etc.

Functions	Ecosystems processes and components	Goods and Services (examples)
Production Functions	***Provision of natural resources***	
14. Food	Conservation of solar energy into edible plants and animals	14.1 Building & Manufacturing (e.g. lumber, skins) 14.2 Fuel and energy (e.g. fuel wood, organic matter) 14.3 Fodder and fertilizer (e.g krill, leaves. Litter)
15. Raw materials	Conversion of solar energy into biomass for human construction and other uses	15.1 Improve crop resistance to pathogens & pests. 15.2 Other applications (e.g. health care)
16. Genetic resources	Genetic material and evolution in wild plants and animals	16.1 Drugs and pharmaceuticals. 16.2 Chemical models & tools. 16.3 Test- and essay organisms
17. Medicinal resources	Variety in (bio)chemical substances in, and other medicinal uses of, natural biota	Resources for fashion, handicraft, jewellery, pets, worship, decoration & souvenirs (e.g. furs, feathers, ivory, orchids, butterflies, aquarium fish, shells, etc.)
18. Ornamental resources	Variety of biota in natural ecosystems with (potential)ornamental use	
Information Functions	***Providing opportunities for cognitive development***	
19. Aesthetic information	Attractive landscape features	Enjoyment of scenery (scenic roads, housing, etc.)
20. Recreation	Variety in landscapes with (potential) recreational uses	Travel to natural ecosystems for eco-tourism, outdoor sports, etc.
21. Cultural and artistic information	Variety in natural features with cultural and artistic value	Use of nature as motive in books, film, painting, folklore, national symbols, architect, advertising, etc.
22. Spiritual and historic information	Variety in natural features with spiritual and historic value	Use of nature for religious or historic purposes (i.e. heritage value of natural ecosystems and features)
23. Science and education	Variety in nature with scientific and educational value	Use of natural systems for school excursions, etc. Use of nature for scientific research

2.3 Impact Model

The impact model component of the approach articulates the specific on-farm (private) and off-farm (social) impacts to ecosystem services and human wellbeing that result from a change in the indicator. For this project, these were identified for each indicator using the ecosystem services and human wellbeing framework presented in Table 2-1. Once the specific impact pathways were identified, dose-response or damage function was identified from a review of the Canadian and international valuation literature. These functions relate the change in the state of the ecosystem to a change in state of ecosystem service or human wellbeing. In the case of SO_2 emissions, this would be a quantitative relationship between urban SO_2 concentrations and the occurrence of asthma.

2.4 Valuation Model

The valuation model describes the economic value associated with a change in the state of an ecosystem service or in human wellbeing. We reviewed the literature to identify data available in relation to economic benefits and costs relevant to the impact pathways of the five focus indicators.

There is a variety of methods whereby the economic values cited in the literature are determined, and these methods were reviewed in Paper # 1 of the joint AAFC-IISD work agreement (IISD 2004). A brief review of this information is provided below for sake of clarity.

There are a variety of terms and concepts that are commonly used in discussions of the non-market value of an activity. The most common examples relate to negative environmental externalities – if a factory or a farm pollutes a river, but does not pay any cost as a result, there is an externality. The polluter can sell its product at a price that does not include the cost of the pollution. That cost is borne by those downstream of the polluter, who either put up with dirty water, or pay to clean it up. The costs of this sort of externality can be calculated, if some data and conceptual difficulties can be dealt with (Venema and Barg 2003).

But there is a broader conceptual framework, into which environmental externalities can be placed. The broad framework or all-encompassing concept can be called "total value", or Total Economic Value" (TEV) (Pearce 1993; Bateman et al. 2003). Pearce breaks TEV down into use and non-use values, in the following categories:

Use Values:
- Direct use value: The value of the use of the resource, for whatever purpose. Agricultural land can produce crops, but it can also provide biomass for energy generation, perhaps forage for animals, and so on. Some of these values will not be easy to quantify.

- Indirect use value: These correspond to "ecological functions", such as protecting watersheds from siltation, or maintaining bio diversity. Carbon sequestration would be an indirect use value, until there is a market for it in a trading system – at which point sequestration will become a direct value.
- Option values: These are also direct values, even though they do not require that there be any specific use of the item at this time. Option values are those that individuals are willing to pay for maintaining the availability of something for their future use, even though the individual has not and may never see it. Old growth forests in British Columbia might be an example.

Non-use values:

- Existence value: This is an indirect value, in contrast to the categories listed above. It is the result of people's willingness to pay for something with no expectation that they themselves will benefit from it. People contribute to organizations to save the Amazonian rain forest or gorillas in Africa, because they feel that these natural wonders should not be destroyed.

The sum of these categories gives TEV. But these are the "economic" values, which is necessarily an anthropocentric calculation. There is a category of non-economic values as well, often called intrinsic values. These values do not depend on human willingness to pay for them, but are intrinsic to the animal, ecosystem, or other part of nature.

A slightly more detailed breakdown of total economic value is given by Bateman et al. (2003). They add the concept of bequest value, which incorporates the value of an environmental good to include the value to those alive now of leaving the good for future generations. This then shows up as both a use value, and as a non use value, on the basis that the future generations will get both from the asset. The diagram below shows the various components of environmental value.

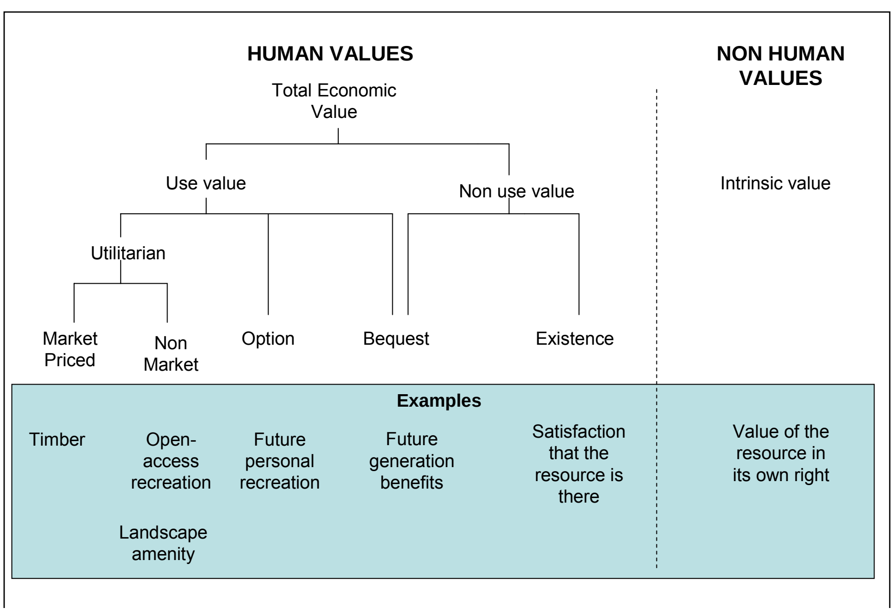

Figure 2-2. Environmental Value (Bateman et al. 2003)

There is another feature of the natural world that TEV and the above diagram do not capture, according to Pearce (1993). That is the fact that the above listing of economic values does not include the value of the system as a whole. He calls them “system characteristics”. The topic is discussed at length by Bockstael et al (2000), who point out that the calculation of economic values as outlined by Pearce is done by measuring a change in value from one specified state to another, and that both states have to be feasible and comprehensible to individuals for the valuation calculation to have meaning.

Using the typology of King and Mazotta (2004), the various approaches to valuation that have been used to date, for circumstances where markets do not directly capture social value are divided into three broad categories. The first is referred to as ***market prices and revealed willingness to pay***, which include prices directly set in markets, as well as prices that can be inferred from market prices. Methods include:

- ***Direct estimation of producer and consumer surplus*** - can be done for markets where there is a reasonable amount of data and supply and demand curves can be calculated
- ***Productivity method*** - Here, the ecosystem value being calculated is one input to a marketed product, so it is necessary to estimate the value of the input as a portion of the value of the marketed product. For example, an increase in the quality of water in a river will decrease the costs of treatment at a municipal treatment plant, thus contributing to an overall cost savings for drinking water users.
- ***Hedonic pricing method*** - can be used to estimate the values of changes in the characteristics of a good. For example, the value that people derive from a nice view from their house can be estimated from data on the cost of houses both with and without a view. The same methodology can be used to value (or derive costs for) such things as air pollution or noise.
- ***Travel cost method*** - is best suited to valuing ecosystems or sites that are used for recreation. Basically, the approach uses the costs that people incur in visiting a place as an indicator of its value.

The second category is ***circumstantial evidence and imputed willingness to pay,*** for example the amount that people are willing to pay to avoid floods can suggest the value of wetlands that will perform this service. The specific methods in this category include ***damage cost avoided, replacement cost, and substitute cost methods.*** These methods estimate ecosystem costs by estimating the cost of damages due to lost services, the cost of replacing services, or the cost of substituting for such services. For example, the damage that might be caused by flooding after the removal of a wetland can be estimated by looking at the area or property that might be flooded, and the cost of replacing the flood control capacity of the wetland can be estimated from engineering estimates of other sorts of control systems.

The third and final category of valuation methods is ***Surveys***, which capture people’s statements of their willingness to pay. The types of survey methods include:

- ***Contingent valuation method*** - The method involves direct surveys of individuals, asking them what they would be willing to pay for certain specific environmental services. The word "contingent" refers to the fact that people are asked how much they would pay for something like an environmental service, contingent on a specific scenario and description of the service. While the methods discussed above try to derive values from market behaviour and engineering cost calculations, CV depends on what people say they would pay for something. The results are controversial, because it is easy to argue that what people say, and what they might actually do, is different. However, such studies are the only way to get some sort of estimates of non use values.
- ***Contingent choice method*** - In this case, the survey does not ask for specific values, but inquires about the choices or tradeoffs that people might make, and infers value figures from this information. The survey will define two or more outcomes including their costs and benefits, and ask the respondents to rank the outcomes.

Benefit transfer is another type of valuation methodology. Benefit transfer provides a methodology by which valuations (of the types described above) obtained in one study can be used elsewhere, in situations shown to be similar enough that such a transfer is reasonable. This depends on whether the services being valued are comparable to the services in the existing study, in terms of the features and qualities of sites and ecosystems, and in terms of the existence of substitutes.

The impact pathway analysis and data review for the five focus indicators are presented in the sections that follow.

2.5 Literature Review for Data and Methods

Information related to details of the agri-environmental indicators was obtained from the 2000 Report of the Agri-Environmental Indicators Project (McRae et al 2000) and the 2004 proposals for improvements to the agri-environmental indicators. This information was supplemented with personal interviews with the indicator experts for each agri-environmental indicator.

Information related to data and methods relevant to the transport, impact and valuation models for each of the focus indicators was obtained from a variety of sources. The primary information source was the Environment Canada's Environmental Valuation Resource Inventory (EVRI).[2] EVRI is "is a searchable storehouse of empirical studies on the economic value of environmental benefits and human health effects. It has been developed as a tool to help policy analysts use the benefits transfer approach."

Information was also obtained from the New South Wales Environmental Protection Authority's ENVALUE database[3]. ENVALUE provides access to Australian and

[2] Available at http://www.evri.ca/.
[3] Available at http://www.environment.nsw.gov.au/envalue/

international “data on environmental values from more than 400 studies covering air, water and land quality; avoidance of noise and radiation exposure; and recreation and other values for natural areas.” The database is described as a “core element of the EPA's ongoing mission to encourage the use of environmental values in decision making.”

Data and methods were also reviewed using standard Internet searches and the services of IISD’s Information Resource Centre.

3. Impact Pathway Analysis for Changes in the Risk of Water Erosion

The impact pathway analysis for changes in the *Risk of Water Erosion* indicator is presented below. Background information on the indicator is provided in Sections 3.1 in relation to the key issue addressed by the indicator and the calculation methods and limitations. The impact pathway analysis, including the pathway, transport, impact and valuation models, is presented in Section 3.2. Section 3.3 highlights some of the main methodological and data gaps observed along with recommendations on preferred methodologies.

3.1. Indicator Overview

The Risk of Water Erosion indicator was created to monitor the extent of cultivated land at risk to erosion by water. This indicator is one of six soil quality indicators – the others being risk of wind erosion, tillage erosion, soil compaction and salinization, as well as soil organic carbon. Water erosion rates are typically highest during heavy summer rain storms and during spring snow melt. McRae et al. (2000) notes that changes in the risk of water erosion over time will primarily be in response to changes in farm management practices. This is an important indicator as increases in water erosion have both on-farm and off-farm impacts – on-farm in terms of a loss in soil productivity, and off-farm in terms of a degradation of water quality in nearby waterways. Therefore, decreases in water erosion bring benefits both on and off the farm.

This risk of water erosion is expressed in the following five classes:

- tolerable (less than 6 tonnes per hectare per year),
- low (6 to 11 t/ha/yr),
- moderate (11 to 22 t/ha/yr),
- high (22 to 33 t/ha/yr), and
- severe (greater than 33 t/ha/yr).

The tolerable class is considered able to sustain long term crop production. All other classes indicate unsustainable conditions and "for which soil conservation practices are needed to support crop production over the long term."

The indicator was estimated using the Revised Universal Soil Loss Equation (RUSLE). RUSLE is a peer-reviewed method that has "withstood the test of time through many applications world wide." Using this method, the average annual soil loss in t/ha/yr is calculated by the equation:

$A = RKLSCP$

The variables for the soil loss equation are as follows:

Parameter	Calculation Method and Data Source
R = a rainfall and runoff factor	Data from 700 weather stations used to calculate R for each SLC polygon. Two different empirical relationships were used for calculating rainstorm and spring snowmelt runoff.
K= a soil erodibility factor	A measure of the inherent resistance to erosion. K factors were determined for each of the dominant and subdominant soil types in each Soil Landscape Polygon (SLC) polygon. K is a function of particle size, organic matter content, structure of the surface soil, and permeability of the soil profile.
L = a slope length factor S = a slope steepness factor	An LS factor was determined for each SLC polygon based on the mapped slope class and surface form.
C = a crop cover and management factor	C was determined from Census of Agriculture crop data reformatted into SLC polygon boundaries. Land use cover maps were used to determine the total agricultural area of an SLC polygon and used to proportion each crop and tillage category.
P = a conservation support practice factor	This data was not available in Census of Agriculture data. Therefore, the P value was set at 1.0.

This methodology is considered adequate for calculation of the risk of water erosion indicator, and therefore, no changes have been proposed for the 2005 reporting period. The only substantive change that has been made is in the data processing methodology. Previous decentralized spreadsheet calculations have now been centralized using ORACLE software (personal communication, Van Vliet 2004).

The indicator was calculated for each of the Census of Agriculture years (1981, 1991, 1996). The map-form presentation of 1996 data for the Prairies is shown on Figure 3-1. The indicator data is further presented in two formats. The first format, shown in Table 3-1, presents the share of cropland in various risk classes for each province. The second format presents graphically the change in area of cropland at risk of tolerable level of water erosion between the 1981 and 1996 Census periods.

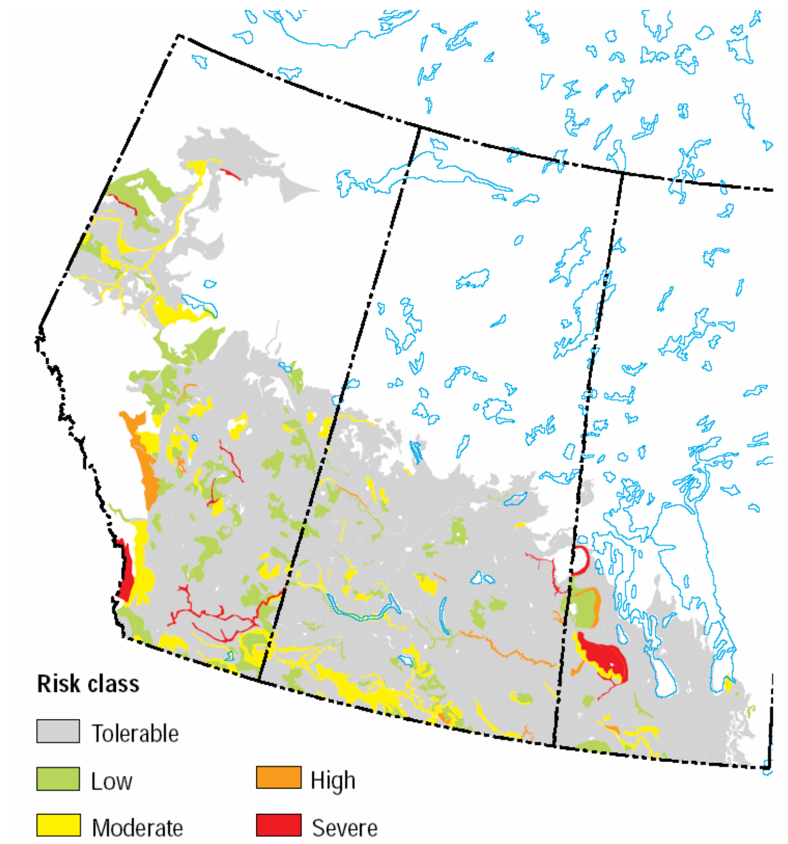

Figure 3-1. 1996 risk of water erosion in the prairies (from McRae et al. 2000).

Table 3-1. Share of cropland in various risk categories (from McRae et al. 2000).

Province	Cropland area* (million ha)	Share (%) of cropland in various risk classes														
		Tolerable			Low			Moderate			High			Severe		
		1981	1991	1996	1981	1991	1996	1981	1991	1996	1981	1991	1996	1981	1991	1996
British Columbia	0.52	56	59	56	25	22	19	12	13	19	5	4	5	2	2	1
Alberta	10.6	75	80	83	15	11	11	8	7	6	2	1	1	<1	<1	<1
Saskatchewan	18.8	64	72	90	24	19	5	7	5	5	4	4	1	2	1	<1
Manitoba	4.9	88	87	89	5	4	4	3	4	4	1	1	1	3	2	2
Ontario	3.4	51	56	58	26	23	27	13	11	6	10	10	10	<1	<1	<1
Quebec	1.6	89	89	88	7	8	9	4	3	3	0	0	0	0	0	0
New Brunswick	0.1	43	45	48	23	32	30	22	14	14	6	6	5	6	3	3
Nova Scotia	0.1	74	71	72	14	15	15	10	12	10	<1	<1	<1	2	3	2
Prince Edward Island	0.1	59	60	59	23	22	23	14	15	19	4	4	0	<1	<1	0

*Cropland area is an average of values for 1981, 1991, and 1996

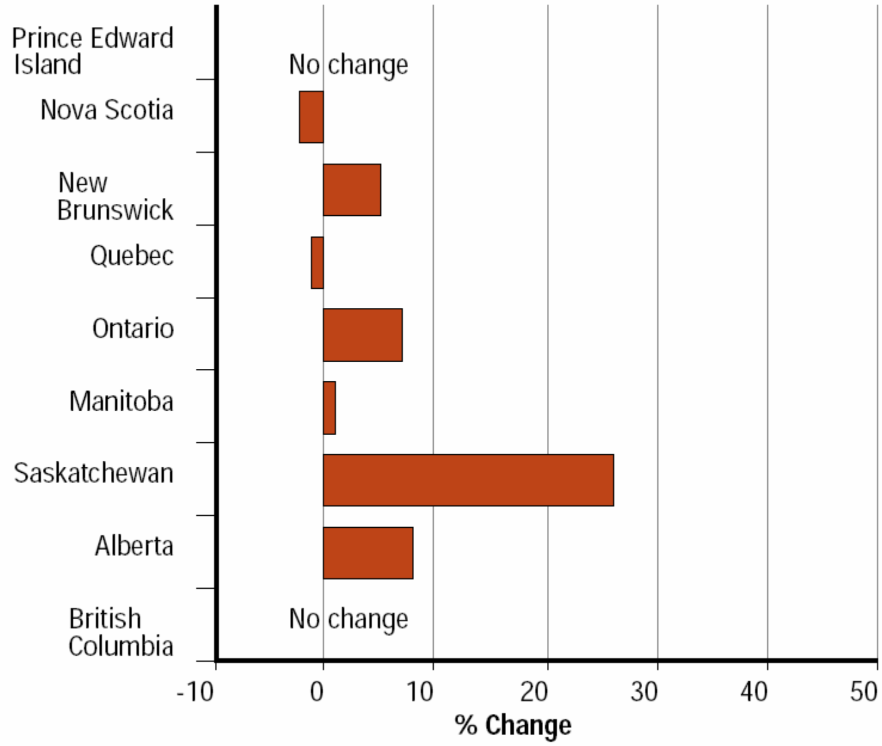

Figure 3-2. Change in area of cropland in the tolerable risk category (from McRae et al. 2000).

McRae et al. (2000) notes that this indicator is currently subject to the following limitations:

- Calculations did not account for improvements resulting from the use of erosion control practices such as grassed waterways terracing, contour cultivation, strip cropping, and winter cover crops.
- Census data are not detailed enough to adequately reflect the geographic distribution of management practices in landscapes where farmland is fragmented, and some calculation errors may occur
- The indicator is based on long term average annual rainfall data that may not reflect single high intensity rainfall events that can cause significant soil erosion.

3.2 Impact Pathway Analysis

The potential impacts that a change in the risk of water erosion indicator could have on ecosystem services were analysed. This analysis is summarized in Table 3-2 and identifies thirteen potential impacts. For these potential impacts, we identified three primary impact pathways:

1. ***Sediment and Turbidity loading effects:*** As a result of water erosion, soil is carried in runoff to agricultural drains and other waterways, where it can contribute to the sediment load on various types of hydraulic structures and waterways (McRae et al. 2000). Sediment loading can effect the: a) performance of drainage, irrigation and navigation and flood control structures; b) habitat of commercially harvested species, aquaculture; c) recreational use of water bodies; and d) cultural use of water bodies.

   Additionally water quality in nearby water bodies can be impacted as suspended soil particles increase the turbidity (cloudiness) of the water. The specific impacts of turbidity on ecosystem services and human wellbeing are similar to sediment loading with the exception that turbidity is less an issue for drainage, irrigation and flood control structures, and more an issue for consumptive water supplies.

2. ***Nutrient and other contaminant loading effects:*** Similar to sediment and turbidity loading, nutrients such as nitrogen and phosphorous and other constituents including pesticides and pathogens are also transported by water erosion to off-farm waterways. The impacts on ecosystem services and human wellbeing are similar in nature to turbidity loading.

3. ***Soil loss effects:*** When soils are eroded, on-site they can become: a) degraded (lower in organic matter and nutrients, less permeable to water, poorly aerated, more difficult to till, more resistant to the penetration of crop roots, increasingly vulnerable to further erosion, and unable to sustain high-yielding crops or agricultural production over the long-term; and b) reformed and redistributed in the landscape.

The first two pathways represent external off-farm social costs, while the fourth is a direct private on-farm cost to the producer. These pathways are depicted on Figure 3-3. The sediment and turbidity loading and the soil loss constituent pathways are discussed in more detail below. The phosphorous constituent pathway is analyzed separately in Section 4.

Table 3-2. Summary of potential impact pathways for changes in the risk of water erosion.

	NAHARP Agri-environmental Indicators
Ecosystem Goods & Services	Risk of Water Erosion
Regulation Functions	
1. Gas regulation	
2. Climate regulation	
3. Disturbance prevention	Sediment loading and overland waterflow effects on flood control;
4. Water regulation	Sediment loading effects on drainage, irrigation and navigation
5. Water supply	Turbidity, P, N, others, effects on consumptive water supplies
6. Soil retention	
7. Soil formation	Overland flow effects on formation of new soils
8. Nutrient regulation	
9. Waste treatment	
10. Pollination	
11. Biological control	
Habitat Functions	
12. Refugium function	Sediment, turbidity, P, N, others, effects on habitat for species
13. Nursery function	Sediment, turbidity, P, N, others, effects on habitat for species
Production Functions	
14. Food	Soil loss effects on crop production
15. Raw materials	
16. Genetic resources	
17. Medicinal resources	
18. Ornamental resources	
Information Functions	
19. Aesthetic information	Erosion effects on aesthetic value of eroded land
20. Recreation	Sediment, turbidity, P, N , other effects on recreational value of water bodies
21. Cultural and artistic information	Effects on use of nature as motive in books, file, painting, advertising, etc.
22. Spiritual and historic information	
23. Science and education	
Human Wellbeing	
Security	
Ability to live in an environmentally clean and safe shelter	
Ability to reduce vulnerability to ecological shocks and stress	Sediment loading and overland waterflow effects on flood control;
Basice material for a good life	
Ability to access resources to earn income and gain livelihood	Sediment, P, N others, loading effects on recreational, tourism, and other cultural information uses
Health	
Ability to be adequately nourished	
Ability to be free from avoidable diseases	
Ability to have adequate and clean drinking water	P, N, other pathogen loading effects on quality of consumptive and recreational water
Ability to have clean air	
Ability to have energy to keep warm and cool	
Good social relations	
Opportunity to express aesthetic and recreational values associated with ecosystems	Effects on use of nature as motive in books, file, painting, advertising, etc.
Opportunity to express cultural and spiritual values associed with ecosystems	
Opportunity to observe, study and learn about ecosystems	

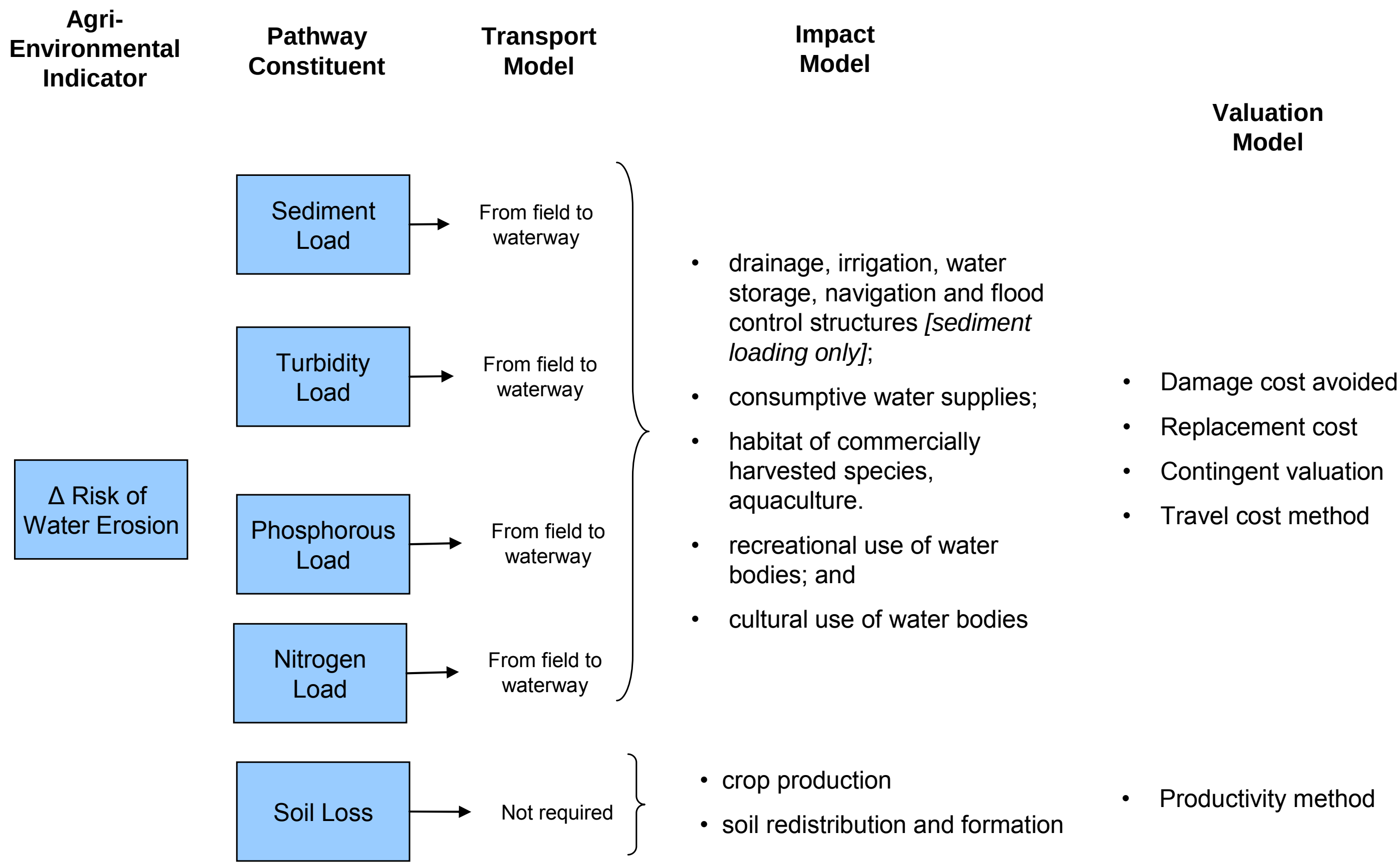

Figure 3-3. Constituent pathways for changes in the risk of water erosion indicator.

3.2.1 Sediment and Turbidity Load Pathway

Transport Model

Valuation of sediment and turbidity loading impacts require quantification of the change in state of off-farm waterways. Form this perspective, the *risk of water erosion* indicator is a pressure indicator on the state of nearby water quality. Determination of water quality in off-farm waterways will therefore require some additional calculations to get from a change in pressure (risk of water erosion in a producer's field) to the change in state in water for a nearby waterway, before a valuation can be made on the impact to specific ecosystem services and human wellbeing. Based on the valuation literature, these calculations are either empirically-based or physically-based.

Empirically-based Approach

Ribaudo (1989) describes an empirically-based transport model in a United States Department of Agriculture study on the offsite benefits of soil erosion control in the United States. In this study, before modelling off-farm transport, estimates of soil erosion by water were made using the Universal Soil Loss Equation for lands enrolled in the Conservation Reserve Program. This information was used to determined the reductions in erosion for each of the 99 Aggregated Sub Areas (ASAs) in the continental U.S. (e.g., the basins of major rivers for which there is data on the discharge of sediments, nitrogen and phosphorous).

The ***transport*** of sediments off-farm and into waterways was determined by multiplying a *sediment delivery ratio* determined for each ASA by the change in soil erosion. The sediment delivery ratio is the ratio of the amount of sediment discharged from a watershed basin into waterways relative to the total amount of soil erosion in a basin. The sediment delivery ratio is a function of stream density and soil type. Data on the sediment delivery ratios were obtained from a study conducted by Resources for the Future (Gianessi et al. 1985) and then applied to the water erosion data (Ribaudo 1989).

Ouyang and Bartholic (1997) note the following with regard to sediment delivery ratios (SDR):

> *There is no precise procedure to estimate SDR, although the USDA has published a handbook in which the SDR is related to drainage area. SDR can be affected by a number of factors including sediment source, texture, nearness to the main stream, channel density, basin area, slope, length, land use/land cover, and rainfall-runoff factors…A small watershed with a higher channel density has a higher sediment delivery ratio compared to a large watershed with a low channel density. A watershed with steep slopes has a higher sediment delivery ratio than a watershed with flat and wide valleys. In order to estimate sediment delivery ratios, the size of the area of interest should also be defined. In general, the larger the area size, the lower the sediment delivery ratio.*

The transport model in the Ribaudo study employed an additional component– a statistical regression model – to relate the sediment discharge levels to sediment concentrations in a waterway, necessary in their study to quantify impacts to recreation and consumptive water use. A log-linear relationship was used to relate sediment concentration to the explanatory variables of material discharge, stream flow and water storage (Equation 1). Data from the U.S. Geological Survey's National Stream Quality Monitoring Network was used. Mean daily flow at the outlet of each agricultural sub area, and water storage volumes were determined using information obtained from the U.S. Water Resources Council. A water storage component was included to account for the settling of particles, and therefore, is inversely related to sediment concentration. Ordinary least-squares regression was used to estimate the coefficients. Ribaudo (1989) notes that "despite the use of highly aggregated data, the relationship between pollutant discharge and concentrations can still be observed."

$$Y = a\, X_1^{a1}\, X_2^{a2} X_3^{a3} \qquad [1]$$

Where:

Y = material concentration (mg/l)
X_1 = material discharge (weight/year)
X_2 = stream flow (volume/day)
X_3 = water storage (volume)

For the Ribaudo (1989) study the regression was as follows:

$$Y = 3.27\, X_1^{-0.88}\, X_2^{-0.40} X_3^{-0.08}$$

Holmes (1988) in a study on the offsite impacts of soil erosion on the water treatment industry used a similar regression, but with the following parameters for the TSS concentration function.

$$\ln Q = -3.71 + 0.72 \ln X_1 + 0.03 \ln X_2 - 0.23 \ln X_3$$

Table 3-3 below summarizes the transport model approaches used in Ribaudo (1989) study for the different impact pathways associated with sediment and turbidity loading.

Table 3-3. Transport Model Approach for Conservation Reserve Program Study

Impact Pathways	**Transport Model**	**Reference**
Drainage structures	▪ 1 to 1 (% change in erosion equal to % change in sediment load)	▪ Ribaudo (1989)
Irrigation structures	▪ 1 to 1 (% change in erosion equal to % change in sediment load)	▪ Ribaudo (1989)
Water Storage structures	▪ 1 to 1 (% change in erosion equal to % change in sediment load)	▪ Ribaudo (1989)

Impact Pathways	Transport Model	Reference
Navigation structures	▪ 1 to 1 (% change in erosion equal to % change in sediment load)	▪ Ribaudo (1989)
Flood control structures	▪ 1 to 1 (% change in erosion equal to % change in sediment load)	▪ Ribaudo (1989)
Recreation	▪ Sediment delivery ratio ▪ Regression to get concentrations	▪ Gianessi et al. (1985), in Ribaudo (1989) ▪ Ribaudo (1989)
Commercial fishing	▪ Sediment delivery ratio ▪ Regression to get TSS concentrations	▪ RRF (1985), in Ribaudo (1989) ▪ Ribaudo (1989)
Consumptive use	▪ Sediment delivery ratio ▪ Regression to get TSS concentrations	▪ Gianessi et al. (1985), in Ribaudo (1989) ▪ Ribaudo (1989)
Industrial Use	▪ Sediment delivery ratio ▪ Regression to get concentrations	▪ Gianessi et al. (1985), in Ribaudo (1989) ▪ Ribaudo (1989)
Cultural Use	▪ Sediment delivery ratio ▪ Regression to get concentrations	▪ RRF (1985), in Ribaudo (1989) ▪ Ribaudo (1989)

McRae (2000) estimated flows of soils from cropland to water for the Grand River Basin in Ontario also using a watershed-based approach. In his approach, the Soil Landscape of Canada (SLC) polygons located within the watershed basin were identified and the total soil deposition into the waters of the watershed calculated as follows:

Y_{GR} = tonnes of soil particles deposited annually into waters of watershed
= $\sum Y_{GR\ SLC}$

$Y_{GR\ SLC}$ = tones/yr of soil loss into study area for each SLC
= $(Y_{SLC\ ty\text{-}1})$(% of SLC lying within study area)

$Y_{SLC\ ty\text{-}1}$ = tonnes/yr of soil loss from land under row crops in each SLC
= (Y)(ha row crop in each SLC)(0.001)

Y = kg/ha/yr of soil loss in each SLC

In determining the soil loss in each SLC (Y), McRae used the method of Wall et al. (1982). Wall et al.'s approach also used a regression equation that was developed using watershed variables – the final regression was as follows:

Y = soil loss into water (kg/ha/yr) = -204+7.9(% row crop)+11.0(% clay)

McRae used the Wall et al. method for three reasons, namely: the regression is specific to southern Ontario; it could be applied at a broad spatial scale; and the required data could be acquired from the AAFC's national soils database. McRae discovered in using these equations that the SLC boundaries do not correspond to watershed boundaries. He dealt with this mismatch by using the ARC/INFO GIS Union function to overlay the relevant portion of a watershed boundary with a land cover file from version 1.9 of the SLC database, and then using the ARC/INFO frequency function to identify the proportion of SLCs falling partially or wholly within the watershed.

Physically-based Approaches

Physically-based approaches have also been used to determine sediment loading rates to waterways. Fox and Dickson (1990) report on the economics of erosion and sediment control in Southwestern Ontario. They used the *Guelph model for evaluating the effects of Agricultural Management systems on erosion and sedimentation* (GAMES) reported by Cook et al. (1985), to estimate the proportion of eroded sediment in a watershed that is discharged to a stream. The GAMES model divides a watershed into field-sized cells with homogeneous features, such as land use, slope and soil characteristics. The Universal Soil Loss Equation is used to estimate the erosion within the watershed and the model calculates the proportion of the mobile sediment that actually enters the stream, net of sediment that is deposited on land in the watershed.

McRae (2000) reported on other uses of the GAMES model by Fox et al. (1995) and van Vuuren et al. (1997), but noted that application in his study was impractical due to the data requirements of so large a study area.

A number of physically-based sediment loading models are reviewed by Donigian and Huber (1991) for an EPA report on modelling of non-point source water quality in urban and non-urban areas.

Impact and Valuation Model

The impact model in this report describes the relationship between the change in the state of the ecosystem (e.g., increase in sediment concentration in a waterway, determined in the transport model) and the resulting impact on the use of a particular ecosystem service or human wellbeing. The valuation model is the method used to relate an economic value to the change in the level of ecosystem service or human wellbeing. The paragraphs below describe the impact and valuation models for the various ecosystem and man-made services impacted by sediment loading.

Drainage services

Ribaudo (1989) developed a damage function for the impact of soil erosion on roadside ditches – sediments carried off farm can fill roadside ditches and increase the potential for the flooding of roads. Ditch cleaning and road services were assumed to be perfect substitutes for road services. The damage function was calculated by obtaining data on ditch maintenance costs from State highway departments. The total State sediment removal costs were specified as a function of gross erosion (i.e., a transport model to describe the movement of sediments to ditches was not determined), rural road mileage and the cost of removing a cubic yard of sediment. The damage function was estimated at $79US/1,000 tons of gross erosion. The author believed this estimate underestimated the damage function because ditch maintenance was likely an imperfect substitute for road service which is the actual use being impacted. The valuation model as summarized by McRae (2000) is presented in Table 3-4.

Table 3-4. Summary of Ribaudo (1989) Valuation Models (from McRae 2000).

Annual offsite damage from soil particles, million 1986 dollars						
Damage category	**Lake States, US $**	**$/tonne**		**Northeast States, US $**	**$/tonne**	
		US $	**CDN $**		**US $**	**CDN $**
Freshwater recreation	189.162	2.88	4.00	336.350	3.71	5.16
Water storage	52.666	0.80	1.11	18.729	0.20	0.28
Navigation	29.954	0.45	0.63	65.131	0.72	1.00
Flooding	65.110	0.99	1.38	103.119	1.14	1.58
Roadside ditches	22.584	0.34	0.47	50.017	0.55	0.77
Freshwater commercial fishing	15.424	0.23	0.32	0.220	.001	.003
Municipal water treatment	113.695	1.73	2.40	183.213	2.02	2.81
Municipal & industrial use	176.053	2.67	3.70	193.679	2.14	2.97
Steam power cooling	4.737	0.07	0.10	33.05	0.36	0.51
Total damages	669.384	10.16	14.11	983.508	10.87	15.10

Similarly, Fox and Dickson (1990) calculated a damage function for drainage ditches in southern Ontario. They used annual costs for removing sediments from roadside ditches (reported by the Ontario Ministry of Transportation and Communications at $4.2 million in 1987 dollars) and for and municipal drains (reported by Wall and Dickson (1978) at $11.2 million), and divided this total expenditure by the total improved cropland area in Ontario to arrive at a $3.41 per hectare damage function. They assume that since the amount of sediment in drains is from gross erosion and not only from sediment delivery, that damage to drainage will be in proportion to gross erosion estimates rather than the sediment delivered to waterways. The valuation model for this work as summarized by McRae (2000) is presented in Table 3-5.

Additionally, Ribaudo (1989) highlighted that sediments in streams can increase flooding potential (sediment deposits can raise the stream bed decreasing stream flow capacity) and cause damage when deposited outside of the stream channel during flooding. No impact model relating sediment discharge to flood frequency existed for use in the Ribaudo study; however, information was available on the erosion damages of flooding.

Table 3-5. Benefits from reduced soil losses into water in selected basins of southern Ontario reported by Fox and Dickson (1990) (from McRae 2000)

Table 28: Benefits from reduced soil losses into water in selected basins of southern Ontario				
Watershed	**Beneficiary**	**Annual Value (1987 $)**		**Units**
		Average	**Range**	
Big Creek	Recreational angling	79.4	---	$ / tonne / year
	Water conveyance	2.27	3.41 – 1.14	$ / ha / year
	Municipal (Mun.) water treatment	20.16	---	$ / tonne / year
Newbiggen Creek	Recreational angling	52.93		$ / tonne / year
	Water conveyance	2.29	3.41 – 1.18	$ / ha / year
	Mun. water treatment	13.44	---	$ / tonne / year
Stratford/Avon	Recreational angling	26.47	---	$ / tonne / year
	Water conveyance	2.37	3.41 – 1.28	$ / ha / year
	Mun. water treatment	6.72	---	$ / tonne / year

Irrigation services

Ribaudo (1989) describe a damage function relating sediment loading and build up in canals. Sediment cleanup costs were obtained for each of 10 farm production regions in the U.S. The relationship between soil erosion and sediment cleanup was assumed to be linear (e.g., a percentage decrease in erosion would produce the same percentage decrease in maintenance costs). It is believed that this method underestimates the damage/benefit function because canal maintenance is likely an imperfect substitute for canal services.

Navigation services

Ribaudo (1989) used a linear relationship between soil erosion reduction and annual dredging costs to develop a damage function for navigation services. For example, a percentage reduction in sediment delivery results in the same percentage reduction in dredging costs. The valuation model as summarized by McRae (2000) is presented in Table 3-3.

Consumptive Use

Consumptive use impacts were estimated for 1) increased turbidity and impact on public water treatment; and 2) increased dissolved minerals and salts on municipal and industrial use. For impacts on water treatment in the Ribaudo (1989) study, changes in TSS concentrations were converted to changes in turbidity (in NTUs) using a relationship developed by Helvey, Tiedmann and Anderson (1985), in Ribaudo (1989). A model relating mineral and dissolved salt concentrations to industrial treatment or damage was not available; therefore, a linear model relating impact to valuation was assumed in which a percentage decrease in sediment load in a waterway, results in the same percentage decrease in damage.

Fox and Dickson (1990) researched the operating costs for water treatment plants in southern Ontario and using the sediment delivery for Ontario estimated by Wall et al. (1982), and estimates of the percentage of total treatment costs attributable to cropland erosion from the U.S. (5% of total costs) developed a damage function relating sediment delivery to the cost of water treatment. This damage function was then adjusted for each watershed to account for the probability that sediments from the sub-watersheds travel to the lakes were water treatment occurs.

Recreational Uses

Fox and Dickson (1990) developed a damage function for recreational fishing for the Thames River watershed in southern Ontario. They note that sediment harms fish in lakes and streams by damaging the spawning and feeding grounds and by reducing respiratory efficiency. They used data from the 1983 Erosion and Sedimentation Control Committee which reported sediment factors for the effect of excess sedimentation on fish populations (e.g., certain fish species respond well to water quality improvements, while others are less sensitive, therefore sediment factors differ among species).

Using data on the number of fish caught in the watershed for the different species along with the sediment factors they estimated the total benefits resulting from the elimination of sediment from streams and lakes in southern Ontario. Then using data from Wall et al. (1982) on the total sediment delivery from cropland and stream banks in Ontario, they arrived at a provincial average loss to recreational angling of $52.93 per tonne of sediment per year. The results of their calculation were summarized by McRae (2000) in Table 3-5. In determining the costs specific to their three study watersheds in the Thames River basin, they accounted for the location of the recreational fishing areas relative to the sub-watersheds where the sediments were eroded and delivered from. This was based on the understanding that depending on the particulate side of the sediment, a certain amount of sediment will settle out in the streams before reaching the lakes and streams at the mouth of the watershed where the recreational fishing occurs.

3.2.2 Nutrient and Other Contaminant Loading

Like sediment and turbidity loading, water erosion results in the transport of nutrients and other contaminants such as pesticides, to off-farm waterways which have impact on certain ecosystem services and aspects of human wellbeing. Many of the water erosion studies reviewed deal with sediment loading together with phosphorous and nitrogen loading. For example, in the Ribaudo (1989) study it is noted that "declines in [total nitrogen] and total [phosphorous] attached to sediment were estimated by applying attached pollutant coefficients for [nitrogen] and [phosphorous] on cropland erosion in each aggregated sub area to the reductions in sediment discharge." The pollutant coefficients for nitrogen and phosphorous were determined in a Resources for the Future study (Gianessi et al. 1985). A study conducted by van Vuuren et al. (1997) dealt with sediment and phosphorous loading in an Ontario watershed using indexing procedures.

Valuation of this impact pathway for the *Risk of Water Erosion* indicator has the potential to integrate with separate valuation exercises undertaken for the *Risk of Water Contamination by Nitrogen*, and also for the *Risk of Water Contamination by Phosphorous* indicators. The impact pathway for the Risk of *Water Contamination by Phosphorous* indicator is analysed in detail in Section 4. The potential integration of this indicator pathway analysis with the water erosion indicator is explored in more detail in Section 8.

3.2.3 Soil Productivity

The soil productivity impact pathway is an on-farm private cost of soil erosion by water. There is no transport model requirement for this impact pathway. The risk of soil erosion by water directly produces a soil loss in tonnes/ha/yr. Therefore, unlike the sediment loading pathway in which soil erosion was a pressure on the quality of water in water ways, soil erosion for the soil productivity pathway is a state indicator and can be used directly in an impact and valuation model.

Impact and Valuation Models

Fox and Dickson (1990) in their analysis of the economics of soil erosion in southwestern Ontario used the Soil Conservation Economics (SOILEC) model to translate the erosion rates given by the Universal Soil Loss Equation (USLE) into reductions in soil productivity by estimating changes in the depth of soil horizons and bulk density. The soil data requirements included topsoil depths, USLE soil erosion variables, rates of crop residue production, and soil bulk density. Economic data requirements included commodity yields, costs of production, real commodity prices, real discount rate, rate of technological change, erosion-induced adjustments of yields and costs of production for alternate tillage systems (yield adjustment values for their Ontario location are provided).

The SOILEC model calculates the present value of a stream of net returns and transforms the present value to an annuity. The results for three locations within their study watershed are summarized in Table 3-6. The on-farm costs (last column in Table 3-6) are then determined as the difference in the annuities (2nd last column in Table 3-6) between the alternative and base tillage practices. Fox and Dickson concluded from this analysis that "although erosion rates are reduced with conservation tillage practices, these systems all exhibit a cost to the farmer, since reductions in yields outweigh the savings in costs of production and long-run on-farm benefits of soil conservation."

Table 3-6. Annual on-farm costs of soil conservation practises for three watersheds in southern Ontario (Fox and Dickson 1990).

Watershed	**Rotation-tillage combinations**[a]	**Annual net returns**[b]	**On-farm costs of adoption**
		($/ha/yr)	
Big Creek	CS FPL	218.60	Base
	CS FCH	201.83	16.77
	CS RIPL	184.91	33.69
	CC FPL	151.79	66.81
	CC FCH	134.89	83.71
	CC RIPL	97.72	120.88
Newbiggen Creek	CC FPL	182.21	base
	CC FCH	164.57	17.64
	CC RIPL	125.75	56.46
Stratford/Avon	2C2A FPL	39.24	base
	2C2A FCH	33.42	5.82
	2C2A NT	28.80	10.44
	3C3A FPL	28.77	base
	3C3A FCH	23.17	5.60
	3C3A NT	18.74	10.03

[a]CC = continuous corn
CS = alternating corn and soybeans
2C2A = alternating 2 years corn and 2 years alfalfa
3C3A = alternating 3 years corn and 3 years alfalfa
FPL = fall moldboard ploughing
FCH = fall chisel ploughing
RPL = ridge planting
NT = no-till
[b]The present value of net returns expressed as an annuity.

A study conducted by van Vuuren et al. (1997) used a similar methodology for the Kettle Creek watershed in southwestern Ontario. Using the SOILEC model and site-specific input parameters, van Vuuren et al. (1997) estimated changes in annual net farm income for all farms in the watershed relative to base practices (assuming all farms adopted the tillage practices). Their results are summarized in Table 3-7. The authors conclude from the results that there are several management practices that are profitable for farmers. They note that the management practices leading to the greatest reduction in loadings incorporated hay into the rotation, but that such a rotation practice was unprofitable for farmers.

Table 3-7. On-farm soil productivity costs for different tillage practices the Kettle Creek Watershed in southwestern Ontario from (van Vuuren et al. 1997).

		Change in net farm income	
Crop rotation	**Tillage practice**	**Without buffer**	**With 4.5-metre buffer**
		(dollars)	
Actual[b]	FMP	0	---
Actual[b]	FMP	---	-34,377
C-C	FMP	62,886	27,993
C-C	SMP	87,342	52,248
C-C	FCH	2,446,739	2,392,298
C-C	NT	2,071,750	2,020,384
C-S	FMP	1,499,952	1,453,274
C-S	SMP	1,518,585	1,471,755
C-S	SD	2,037,978	1,986,889
C-C-C-W	FMP	-497,266	-527,567
C-C-C-W	FCH	1,263,547	1,218,808
C-S-W	FCH	293,469	256,685
C-C-H-H-H	FMP	-2,360,561	-2,375,583
H-H	NT	-3,980,580	-3,980,580
C-H	FMP	-1,968,105	-1,986,344
C-H	SMP	-1,928,471	-1,945,919

[a]Measured in 1991 dollars.
[b]36% corn, 27% soybeans, 10% wheat, 4% oats, 18% other crops.
C = corn, S = soybeans, W = wheat, H = hay, SD = spring disc, FMP = fall moldboard plow, SMP = spring moldboard, FCH = fall chisel plow, NT = no-till

Crosson (2003) conducted a global review of the economics of soil erosion and maintaining soil biodiversity in which he considered both on-farm and off-farm costs and benefits. Crosson cites an estimate annual on-farm cost of soil erosion in the U.S. at $100 million per year, or $0.60 per hectare per year. He asserts that because U.S. farmers have strong, enforceable property rights, that they have strong incentive to keep erosion-induced costs within limits acceptable to them, and that farmers in the U.S. have indeed done this. He therefore concludes that no policies are needed in the U.S. to provide incentive to lower soil erosion for purposes of maintaining crop productivity. It is also his judgement that although this does not address inter-generational interests, given the low on-farm costs of erosion, it is almost certain that intergeneration interests would not demand they be lower.

3.3 Discussion

From a methodological perspective the results of the impact pathway analysis are positive in that valuation studies for water erosion have been previously been undertaken in both Canada and the United States. The scale at which these studies were conducted is important to note. The Ribaudo (1989) study was conducted at an aggregated sub area (ASA) level (e.g., the basins of major rivers, of which there are 99 in the continental U.S.) and scaled up to farm production areas (FPAs). This aggregation to the national level was necessary to evaluate the performance of the Conservation Reserve Program. The Canadian studies reviewed were all undertaken at the watershed or basin level (e.g., Fox and Dickson (1990), van Vuuren et al. (1997); McRae (2000).

With regard to the sediment loading pathway, a key methodological gap however, does exist. The impact pathway analysis revealed that although the risk of water erosion indicator does provide soil loss rates on farmland, some additional empirically- or physically-based modelling will be required in order to determine the amount of sediments that are discharged to major waterways. The good news is that a range of methods have been developed and applied at different scales, and therefore it can be done. Exactly which method to proceed with would require additional feasibility analysis. It is recommended that any ground-testing of valuation methods for changes in the water erosion indicator, both empirically-based such as those employed by McRae (2000) and Ribaudo (1989), and physically-based modelling efforts, such as those used by Fox and Dickson (1990) and van Vuuren et al. (1997) be applied. An evaluation can then be made on the most appropriate methodology given the need to apply to watersheds and basins all across Canada in order to be able to aggregate to a national level.

From a data perspective the results of the impact pathway analysis are also positive. Data has been generated through a number of U.S. and Canadian studies, and these studies themselves have used physical and cost data from other sites to conduct valuations (i.e., benefit transfer). It is apparent however, that site-specific data, both physical and economic, will need to be collected for different regions (perhaps by province, eco-zones, or major watersheds).

Regarding the on-farm private costs associated with soil loss, the methodology appears to be well established based on recent literature, and the data requirements not onerous. The method of Fox and Dickson (1990) and van Vuuren et al. (1997) would appear appropriate for such calculations if no other similar method is currently being used by AAFC.

The impact pathway analysis revealed an obvious linkage to two other agri-environmental indicators, namely risk of nitrogen and risk of phosphorous contamination. The phosphorous loading pathway is analysed in detail in Section 4, and statements regarding methodological and data gaps are discussed there.

4. Impact Pathway Analysis for Changes in the Risk of Contamination of Water by Phosphorus

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4.1 Indicator Overview

For several years, elevated phosphorus concentrations in the rivers draining agricultural lands have been problematic. Agriculture seems to be one of the activities which contribute in an important way to the loads of phosphorus in the form of non-point source pollution. Given upstream and downstream transport of the pollutants, it is important to evaluate quantitatively on a watershed basis, the agricultural, economic and environmental impacts of policies or payments on the quality of water; more precisely the intervention programs or management activities which result from this. This quantitative evaluation becomes crucial when we want to determine the contribution of the problematic sub-watersheds and the efficacy of the control interventions based on the conservation of the soil or on the best management practices.

Non-point source phosphorus contamination of surface waters is a very complex set of transfers that involves many processes before phosphorus reaches surface waters. To overcome the limitations of using a soil test phosphorus threshold as the only measure of site phosphorus loss potential, many indicators and models have been developed to help resource managers and stakeholders assessing site vulnerability to phosphorus loss. Those are used as a qualitative approach to evaluate the impacts of agriculture on water quality. The phosphorus indices are designed as a simple, semi-quantitative tool to estimate the risk of phosphorus transfer to surface water from various fields (Lemunyon and Gilbert 1993). The Indicator Of Water Contamination by Phosphorus (IROWC-P) is designed to measure progress in reducing the risk of water contamination by agriculture and where the relative risk of such contamination is higher, and how this risk is changing over time based on the five-year period of data Census frequency (van Bochove et al. 2004).

The relevance of developing a national indicator of risk of water contamination by phosphorus for Canada is to identify critical areas across the country where more research is required to protect surface water (van Bochove et al. 2004). Once identified, these areas would be investigated carefully at the operational management watershed scale.

4.1.1 Origin, development and description

The IROWC-P derives from the P-Index developed by Lemunyon and Gilbert (1993) and USDA-NRCS (1994). This indicator allows the evaluation of various topographic and pedological configurations like various practices of management on the risk of transport of phosphorus towards rivers. The procedure is based on characteristics like erosion, overland water transport (i.e., surface runoff), type of vegetation, presence of pasture, nutrients concentration in soil as well as the amounts, mode and the period of phosphorus application. The modified version of the national IROWC-P is built on the first IROWC-P version developed by Bolinder et al. (2000). The modified version will include a transport – hydrology component that will improve assessment of the risk of phosphorus transport to water by erosion, infiltration and surface runoff to water (van Bochove et al. 2004). Both versions are calculated at the SLC polygon and watershed levels using existing Census of Agriculture, farm environmental management surveys, hydrology and climate databases (van Bochove et al. 2004).

4.1.2 Calculation method

The indicator evaluates the impact of agricultural uses with phosphorus balance, phosphorus soil saturation and movement towards surface water. IROWC-P is expressed in the following five classes of risk of water contamination: very weak low (negligible risk), weak low (acceptable risk), moderate (evaluation of the necessary situation), high (necessary action) and, very high (necessary immediate action). The risk classes of very weak to moderate indicate that the capacity of the environment makes it possible to support sustainable agriculture without major changes of the management practices. On the other hand, the risk classes of high to very high indicate that either production shows a surplus of phosphorus, that the ground is too rich in phosphorus, or that the mechanisms of transport of phosphorus are important.

IROWC-P includes three components: the *soil P-status component* (PS), the *annual P-balance component* (PB) and the *P transport component* (PT). These components will be weighted to estimate their relative importance for phosphorus transfer and rated by their corresponding phosphorus class of risk values. Table 4-1 summarizes the components, their site characteristics and weighting factors. For the modified version of IROWC-P, a hydrology component (infiltration, topographic index, tile drainage, preferential flow and surface drainage density) will be added to the existing phosphorus transport component. A validation and calibration of this modified site characteristic will be done and the weighting factors will therefore change. This new component is under development. The three component values will be combined according to the following equation to estimate the risk of water contamination by phosphorus (van Bochove et al. 2004):

$$\textbf{IROWC-P} = \textbf{(PS + PB) PT}$$

As mentioned, IROWC-P values will be attributed to five risk classes to obtain a corresponding magnitude of risk for each polygon. The following sections present a brief description for each IROWC-P components. For a more detailed description, see van Bochove et al. (2004).

Table 4-1 Phosphorus Indicator for assessing the vulnerability of a site

	Phosphorus loss rating (value)				
Site characteristics (weighting factor)	**Very low (1)**	**Low (2)**	**Medium (4)**	**High (8)**	**Very high (16)**
Phosphorus status					
Phosphorus soil test (STP)[1] (2.5)	< 60	60-150	150-250	250-500	> 500
Degree of soil phosphorus saturation (DSPS)[2] (2.0)	0-2.5%	2.5-5.0%	5.0-10%	10-20%	> 20%
Phosphorus balance					
Mineral fertilizer phosphorus[3] (1.0)	< 50%	50-100%	100-150%	150-200%	> 200%
Manure phosphorus[4] (2.0)	< 50%	50-100%	100-150%	150-200%	> 200%
Crop residues phosphorus[5] (1.0)	< 2%	2-5%	5-20%	20-50%	> 50%
Phosphorus transport[6]					
Soil erosion[7]	*< 500*	*500-2000*	*2000-6000*	*6000-15000*	*>15000*
Overland flow potential	Very low	low	Moderate	High	Very high
Infiltration[8]					
Topographic index[8]					
Tile drainage[8]					
Preferential flow[8]					
Surface drainage density[8]					
Weighted rating values	12-18	19-36	37-72	73-144	145-192
Site vulnerability classes	Very low	Low	Medium	High	Very high

[1] Mehlich-3 extractable P (kg P ha^{-1})
[2] (Mehlich-3 P/Mehlich-3 Al) x 100
[3] Estimated from dollars spent on fertilizer and lime at the polygon level (source: Census of Agriculture database)
[4] Estimated from livestock, manure production coefficients and manure P coefficient for each category (source: Census of Agriculture database)
[5] Estimated for phosphorus uptake and phosphorus harvest coefficients (source: Census of Agriculture database)
[6] Many subcomponents are under development (algorithms, weighted factors and P loss rating value are to be precise.
[7] Soil water erosion loss (kg ha^{-1})
[8] Hydrology component in development for the new version of IROWC-P

Phosphorus status component

The phosphorus status component is characterized by the degree of soil phosphorus saturation and its long-term capacity to retain phosphorus. It is defined as the ratio of soil-test phosphorus to phosphorous sorption capacity, an inherent soil characteristic. The phosphorus soil test has five rating values: very low to very high (Table 4-1). The phosphorus soil analysis is estimated according to soil type and origin (by province) using the following four methods: Mehlich-3 extractable phosphorus, Olsen-P, Kelowna or Bray-P1. Soil phosphorous saturation will be computed at both the SLC and watershed levels using the following two databases: a soil-test P and P sorption capacity. The degree of soil phosphorus saturation is rated from very low to very high in five increments (Table 4-1).

Annual phosphorus balance component

The phosphorus balance component is comprised of three subcomponents (Table 1). *Mineral fertilizer phosphorus* is estimated from crop fertilizer phosphorus recommendation rates using information on the status of current soil-test phosphorus levels. These estimations will be weighed against values derived from provincial summaries (total dollars spent on fertilizers and lime) and on other attributes. For example, m*anure phosphorus* is assessed based on data from the Census of Agriculture, including number of animals in different categories, manure production coefficient and manure phosphorus coefficients for each animal category. *Crop residue phosphorus* is estimated based on the Census of Agriculture database and Provincial Census information: phosphorus uptake and phosphorus harvest coefficients. It is noteworthy that only major annual crops and hay categories are considered. This subcomponent estimates the quantities of exported phosphorus and quantities of crop residue phosphorus remaining on the agricultural soil after harvest.

P transport component

Several factors are used to assess the phosphorus transport component of the modified IROWC-P: particulate phosphorus factors (erosion rate), dissolved phosphorus factors (surface runoff and infiltration) and connectivity factors (topographic index, tile drainage, preferential flow, and surface drainage density) (Table 4-1). As this component is currently under development, weighting values for most subcomponents remain to be estimated.

4.1.3 Interpretation of the results

Risk of water contamination by phosphorous was only calculated for the province of Quebec due to the unavailability of relevant data for the rest of Canada. Data for phosphorous content and degree of phosphorous ground saturation are essential for a calculation of phosphorous status in the ground and for an assessment of phosphorous in agricultural production. (Bolinder et al. 2000). The indicator was calculated for each Census of Agriculture year (1981, 1991, 1996 and 2001). An evaluation of remaining Canadian provinces should be completed for 2008.

The results from indicator calculations are illustrated with polygon maps showing five classes of risk. In addition to identifying polygons or regions at higher risk of water contamination, the maps indicate factors or parameters responsible for the risk. Further, once a map depicting changes in contamination risk classes based on temporal tendency is produced, the evolution of risk on a yearly basis will also be shown[4] (McRae et al. 2000). These maps are useful for assessing the impact of agricultural uses, such as crop type, livestock, soil, fertilization and landscape, on water contamination risk.

4.1.4 Limitations

As mentioned, the indicator was calculated only for Quebec because it is the only database available to calculate phosphorous content and degree of soil phosphorous saturation. Limitations due to the database period (1981-2001) prevented an evaluation of the enrichment of phosphorous content in soil generated by the evolution of agricultural management practices. Consequently, the calculation of the phosphorus balance is related directly to

[4] As of now, only the map for the year 1996, was produced.

phosphorus contents in the ground to estimate the rates of application of phosphate-enriched fertilizers by polygon.

While several management practices may have a significant impact on the calculation of risk, all could not be included in the indicator. For example, adding phytase in some livestock feed could modify the content and the solubility of phosphorus in manure. The Census of Agriculture database, however, does not include manure management and application data. Thus only a gross assessment based on quantity of inorganic fertilizers used, is available at the polygon scale.

The calculation of the risk of phosphorus transfer towards water considers only water erosion, the principal mechanism of phosphorus transport, without evaluating hydrological processes and connectivity of arable lands to the hydrographic network. The estimation does not consider the impact of the snowmelt, nor the climatic variation between years and seasons.

The use of numerous data sources used to calculate the indicator (from 1981 to 2001) and the aggregation of the available data to various scales (unit of census, agricultural areas, municipalities, etc.) at the polygon scale introduce estimation errors. Uncertainty associated with the calculation of risk of water contamination by phosphorus was not quantified and it will be assessed in the modified version of IROWC-P.

4.2 Impact Pathway Analysis

The impact pathway analysis enables us to develop a framework for analysis of the issues involved in estimating the economic impacts of changes in IROWC-P. Pressures driving water contamination arise from both point and non-point sources, and therefore, this is the only pathway constituent for this indicator. Given that the objective of the IROWC-P is to assess the performance of Canadian agriculture, we chose to focus on non-point sources of phosphorus and in particular eutrophication to waterways, which is the principal problem associated with phosphorus contamination.

The potential valuation pathways for changes in risk of water contamination are summarized in Table 4-2. Functional changes include water contamination by phosphorus, leading to eutrophication, effects on habitat functions and on irrigation water, and effects on several information functions (aesthetic and recreational values).

4.2.1 Transport Model

The calculation of IROWC-P provides no quantification of the phosphorus loading factor. It is important to remember that the purpose of the indicator is to give an indication of the risk of water contamination with the help of site vulnerability classes. Thus, as of now, this leaves a gap in the transport model in the impact pathway analysis. Even with this gap, it is still possible to obtain a good qualitative assessment of the mechanism involved in evaluating vulnerability of water contamination by phosphorus. The IROWC-P's site characteristics give a good indication of the P status, P balance and P transport and also indicate which characteristic has the most influence. Therefore, a good understanding of the mechanisms involved in phosphorus water contamination risks should lead to a better understanding of phosphorous behaviour and movement.

If a direct relationship (transport model) between the indicator of risk and the water quality does not currently exist, a correlation will need to be established. In the literature, most studies dealing with the economic assessment of changes in water quality use phosphorous loading or phosphorous concentration to establish an economic relationship to water quality (Table 4-3). Several studies have shown the benefits of improving water quality using a phosphorus concentration reduction assessment following implementation of agricultural best management practices. (Eisen-Hecht and Kramer 2002; Mathews et al. 2002; Loomis 2000; Holmes et al. 1999; van Vuuren et al. 1997; Stonehouse 1999). Bingham et al. (2000) and Legget and Bockstael (2000) also illustrated the benefits of water quality improvement in the case of point source pollution. In both cases, benefits associated with recreational water use and property values were clearly established.

One of the challenges of valuing changes in this indicator is establishing a relationship between the estimated water contamination risk class and water quality. More research is needed to develop an appropriate transport model for this indicator. Research is currently underway to establish a correlation between the IROWC-P value and water quality (Farida Dechmi, personal communication).

Table 4-2 Potential impact pathways for the IROWC-P

	NAHARP Agri-environmental Indicators
Ecosystem Goods & Services	Risk of Water Contamination by Phosphorus
Regulation Functions	
1. Gas regulation	
2. Climate regulation	
3. Disturbance prevention	
4. Water regulation	
5. Water supply	Water contamination by phosphorus
6. Soil retention	
7. Soil formation	
8. Nutrient regulation	Water contamination by phosphorus
9. Waste treatment	Water contamination by phosphorus
10. Pollination	
11. Biological control	Water contamination leading to possibly eutrophication (algal blooms, aquatic species)
Habitat Functions	
12. Refugium function	Effects on habitat for commercially harvested species
13. Nursery function	Effects on habitat for commercially harvested species
Production Functions	
14. Food	Effects on irrigation water
15. Raw materials	
16. Genetic resources	
17. Medicinal resources	
18. Ornamental resources	Effects on irrigation water
Information Functions	
19. Aesthetic information	Effects on aesthetic value of water
20. Recreation	Effects on recreational value of water bodies
21. Cultural and artistic information	Effects on use of nature as motive in books, file, painting, advertising, etc.
22. Spiritual and historic information	
23. Science and education	
Human Wellbeing	
Security	
Ability to live in an environmentally clean and safe shelter	
Ability to reduce vulnerability to ecological shocks and stress	
Basice material for a good life	
Ability to access resources to earn income and gain livelihood	P loading effects on recreational, tourism, and other cultural information uses
Health	
Ability to be adequately nourished	
Ability to be free from avoidable diseases	
Ability to have adequate and clean drinking water	P loading effects on quality of consumptive and recreational water
Ability to have clean air	
Ability to have energy to keep warm and cool	
Good social relations	
Opportunity to express aesthetic and recreational values associated with ecosystems	Effects on use of nature as motive in books, file, painting, advertising, etc.
Opportunity to express cultural and spiritual values associed with ecosystems	
Opportunity to observe, study and learn about ecosystems	

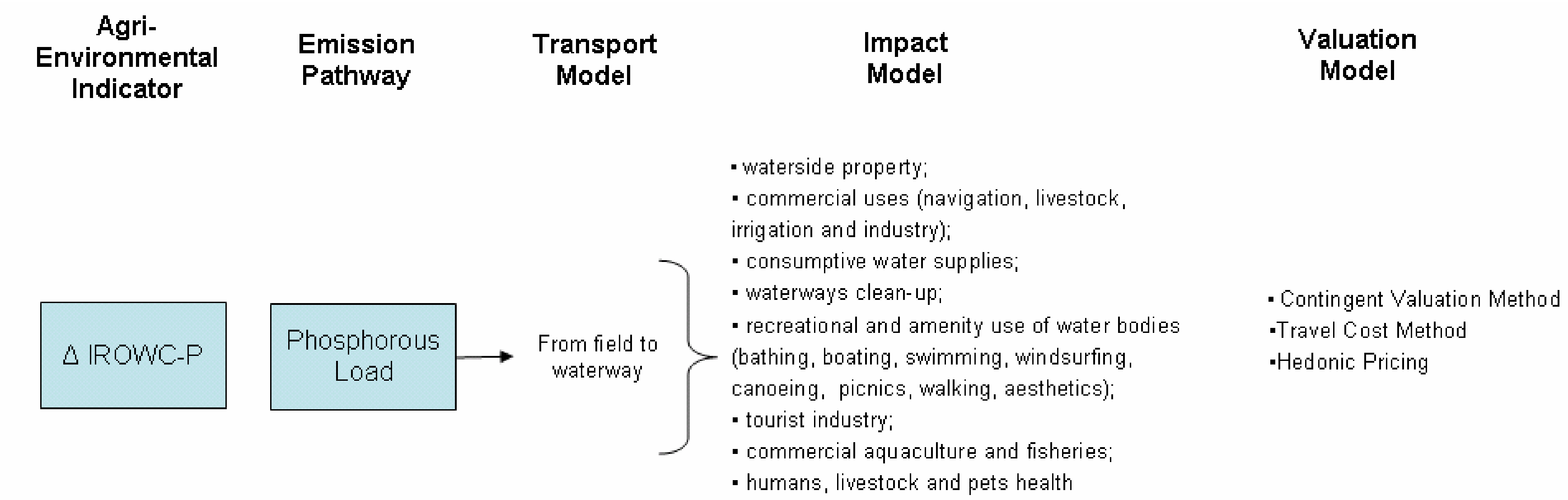

Figure 4-1 Impact pathways identified for the IROWC-P.

Table 4-3 Summary of studies with phosphorus loading or concentration as transport model

Impact pathways	Description	Benefits	Valuation method	Location	Reference
Fishing and transport	Water quality improvement (sediment and phosphorus concentration)	179 000 $CAN and 2,4 M $CAN	Avoid cost	Kettle Creek watershed (Ontario)	van Vuuren *et al.*(1997) Stonehouse (1999)
Recreational activities	Water quality improvement by a reduction of 40% of P levels in the river	142 M $US	Contingent valuation	Minnesota river	Mathews *et al.* (2000) Mathews *et al.* (1999)
Recreational and aesthetic uses	Water quality improvement by a P concentration reduction to reach certain water quality standards	13 and 63 M $US	Benefits transfer	Lake Champlain	Holmes *et al.* (1999)
Recreational activities	Water quality improvement by implementation of several politics and regulations	623 000 $CAN and 17,8 M $CAN	Benefits transfer	Newfoundland	d'Adi Nolan Davis and Gardner Pinfold Consulting (1996)
Consumptive and recreational uses and wildlife	Water quality improvement following the implementation of the *Clean Water Act*	5 and 11 MM $US	Contingent evaluation	United States	Lyon and Farrow (1995)
Recreational uses, commercial fishing, health, intrinsic values and housing	Water quality improvement following the implementation of the *Clean Water Act*	358 M $US and 1,8 MM $US	Benefits transfer	Chesapeake bay	Morgan and Owens (2000)
Recreational uses, fishing habitat, wildlife	Water quality improvement by a P concentration reduction	19 and 71 M $US	Contingent valuation	South Platte watershed (US)	Loomis (2000)

Accordingly, hypotheses concerning the transport model are needed to establish a relationship between water quality improvement and change in water contamination risk. Among the possibilities, the probability of exceeding a given water quality standard could be used to establish when the activity can be practiced (Salvano et al. 2004). The probability is expressed as the number of days water use is possible (Rousseau et al. 2000). Therefore, the latter could be used to assess the benefits associated with water use. As mentioned, the indicator does not provide a quantitative estimation of phosphorus loads; therefore the connection between risk classes and probabilities must be established (Figure 4-2). If water quality standards are chosen as the criteria to assess the transport mode, the frequency of exceeding those criteria could be used to establish the impact of changes in the indicator.. Pretty et al. (2002) used this approach to evaluate the damage costs of eutrophication. In that study, the frequency of eutrophication (algal blooms) were recorded and used in the following equation:

$$f_c = (I_{bg} \text{ x } N)/C \text{ x } (S_{1/2} \text{ or } S_1) \text{ x } Y)$$

f_c = frequency of closure
I_{bg} = number of incidents of algal blooms
C = number of water bodies affected
N = number of days water body closed for each incident
$S_{1/2}$ = season length (days in half year)
S = season length (days in full year)
Y = number of years of data

The use of a similar equation and some regional water quality data could establish the necessary relationship between risk classes and water quality. With those correlations, it could be possible to assess the monetary benefits associated with changes to the indicator. To do so, the probability of frequency of days that water quality has an impact on ecosystem services and goods could be related to risk classes. However this correlation would need to be specific for each studied region since the frequency of days will be regionally established. The following section presents the impact model analysis for IROWC-P.

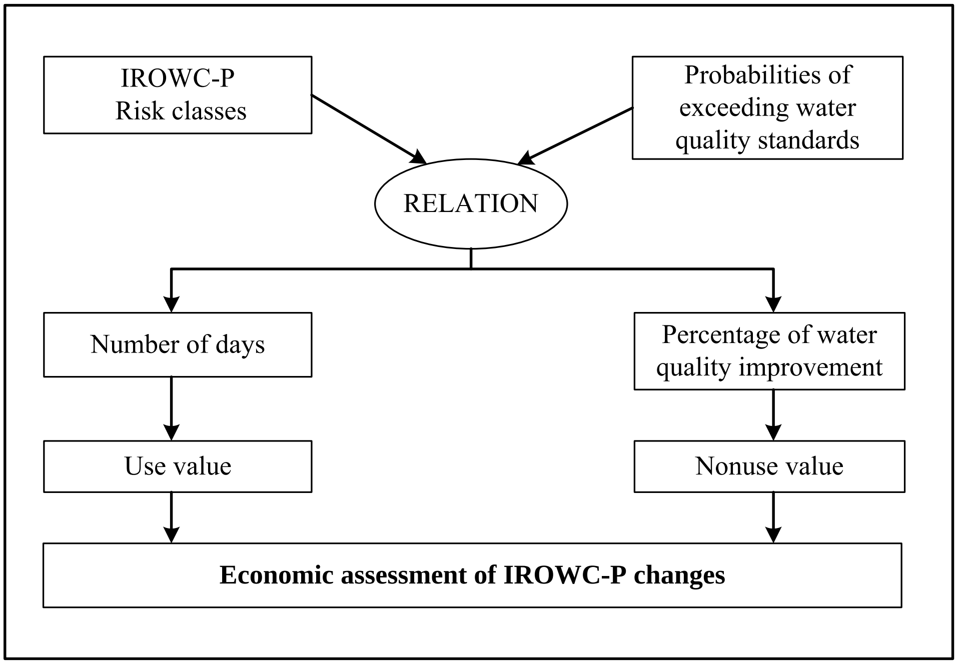

Figure 4-2 Framework for economic assessment of IROCW-P changes.

4.2.2 Impact and Valuation Model

As mentioned, the impact model is the relationship between the change in the state of the ecosystem and the resulting impact on the use of a particular ecosystem or man-made service. Pretty et al. (2003) present two major categories of impact pathways identified by cost category: damage costs (value-loss) and policy response costs. Table 4-4 presents a summary of the annual cost of freshwater eutrophication in the United Kingdom. For the purpose of this study, only the damage costs will be considered.

According to Figure 4-1, eight impact pathways were chosen to perform a more detailed analysis. Those pathways include waterside property, commercial and consumptive uses of water, recreational and amenity uses, commercial aquaculture and fisheries and finally, human, livestock and pet health. A brief review of the literature and studies revealed numerous studies associating monetary benefits with water quality improvement. Table 4-5 presents research that used either phosphorus concentration or phosphorus loading for the transport model in the assessment. Some research also conducted a global analysis. For example, The Value of Natural Capital in Settled Areas of Canada (Olewiler 2004) presented a net value for conservation of natural capital in the upper Assiniboine River Basin ranging between 0.46 and 1.37 \$/hectare/year for water-based recreation and benefits between 2.08 and 6.45 \$/hectare/year for increased wildlife viewing. Also, reports like *The Importance of Nature to Canadians*: *The Economic Significance of Nature-related Activities* (Environment Canada 2000) and *The Importance of Nature to Canadians: Survey Highlights* (Environment Canada 1999) would be important in the determination of the economic benefits associated with water use activities.

Pretty et al. (2002) also prepared an interesting summary of economic valuation studies of water bodies listing consumer surplus and willingness to pay for water-based recreation activities. The results were either presented as a value per household or person per year or per visit. Table 4-5 presents a summary of monetary values associated with water uses. Those values could be very useful for the assessment of water-based activities by a benefit transfer.

Table 4-4 Summary of the annual cost of freshwater eutrophication in the United Kingdom (Pretty et al. 2002)

I. DAMAGE COST *– reduced value of clean or non nutrient-enriched water*

A. Social damage cost

1. Reduced value of waterside housing
2. Reduced value of water bodies for commercial uses (abstraction, navigation, livestock watering, irrigation and industry)
3. Drinking water treatment costs (treatment and action to remove algal toxins and algal decomposition products.
4. Drinking water treatment costs (to remove phosphorus)
5. Clean-up cost of waterways
6. Reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing), angling, and general amenity (picnics, walking, aesthetics)
7. Revenue losses for formal tourist industry
8. Revenue losses for commercial aquaculture, fisheries, and shell-fisheries
9. Health costs to humans, livestock and pets

B. Ecological damage cost

1. Negative ecological effects on biota (arising from changed nutrients, pH, oxygen), resulting in changed species composition (biodiversity) and loss of key sensitive species

II. POLICY RESPONSE COSTS *– costs incurred in responding to eutrophication*

A. Compliance control costs arising from adverse effects of phosphorus enrichment
   1. Sewage treatment costs to remove phosphorus
   2. Costs of treatment of algal blooms and in-water preventative measures
   3. Costs of adopting new farm practices that emit less phosphorus

B. Direct costs incurred by statutory agencies for monitoring, investigating and enforcing solutions to eutrophication
   1. Monitoring costs for water
   2. Cost of developing eutrophication control policies and strategies

Table 4-5 Monetary values associated with water uses

Impact Model	Water use description	Value	Year*	Location	Analysis method**	Study
Recreational and amenity use of water bodies	Swimming	20,50 $US[1]	1990	United States	CV	Bergstrom & Cordell (1991)
	Kayak	17,47 $US[1]			CV	
	Fishing	24,57 $US[1]			CV	
	Picnic	17,21 $US[1]	1997		CD	
	Camping	20,61 $US[1]			TC	
	Biking	15,12 $US[1]			TC	
	Swimming	20 - 32 $US[1]	1997	Oregon (U.S.)	BT	Bingham *et al.* (2000)
	Navigation	22 –43 $US[1]				
	Navigation	24,49 $US[1]	1997	United States	CV	Carson and Mitchell (1993)
	Fishing	20,80 $US[1]				
	Swimming	20,54 $US[1]				
	Navigation	18,92 $	1991	BC (Canada)	CV	Crane Management Consultants (1992)
	Canoe	18,92 $[1]				
	Kayak	18,92 $[1]				
	Swimming	23,91 $[1]				
	Camping	23,91 $[1]				
	Picnic	23,91 $[1]				
	Hiking	24,95 $[1]				
	Swimming	22 $[1]	1991	Great Lakes	BT	Hickling Corporation (1993)
	Navigation	27 $[1]		(Canada)	BT	
	Canoe	35,25 $US[1,2]	1996	Ohio	BT	Hitzhusen et al. (1991)
	Fishing	31,08 $US[1,2]		(United States)		
	Picnic	17,71 $US[1,2]				
	Park	13,62 $US[1,2]				
	Canoe	66,65 $[6]	1993	Ontario (Canada)	CV	Rollins (1997)
	Swimming	19-30 $US[1]	1997	Oregon (U.S.)	TB	Smith et al. (1999)
	Fishing	26 $US[1]	2001	United States		US Fish and Wildlife

Impact Model	Water use description	Value	Year*	Location	Analysis method**	Study
						Service (2001)
	Wildlife viewing	738 \$US[4]				
Consumptive water supplies	Groundwater	634,68 \$US[3]	1996	United States	CV	Crutchfield *et al.* (1997)
	Human consumption	70 \$[4,6,7] 78-90 \$[3,6]	1993	BC (Canada)	CV	Hauser and van Kooten (1993)
Humans, livestock and pets health	Domestic uses	0,16 \$US[5]	1994	United States	BT	Frederick *et al.* (1996)
Commercial use	Navigation	0,12 \$US[5]				

* The year of reference represents the year with which the value is associated.
** CV = Contingent valuation, BT = Benefits transfer, TC = Travel costs
1 Per person per day
2 Expenses
3 Per household per year
4 WTP: Willingness-to-pay
5 Per person per year
6 Per m^3
7 Protection expenses

4.3 Discussion

Indicators of phosphorus contamination are designed as a qualitative approach to evaluate the impact of agriculture on water quality. Development of the Indicator of Water Contamination by Phosphorus at the national level was designed to show where the relative risk of water contamination by agriculture will be higher, and how this risk will change over time. IROWC-P will evaluate the impact of agricultural uses with phosphorus balance, phosphorus soil saturation and movement towards surface water. For the calculation of the IROWC-P at the national level, much work needs to be done. The preparation of appropriate databases and specification of the weighting factors for the site characteristics will need to be established

An impact pathway analysis was performed to help us develop a framework for economic analysis of the issues involved in estimating the economic impacts of changes in IROWC-P. Those changes were largely identified as eutrophication caused by water contamination by phosphorus. The principal regulation functions affected were habitat functions, irrigation water and finally several information functions (aesthetic and recreational values). Based on this impact pathway analysis, the principal gap identified was the phosphorus loading. This gap prevents the direct economic analysis of the changes in the indicator. Most studies involved in economic assessment of water quality used either phosphorus concentration or loading as the variable to show the water quality improvement and then the benefits related to this.

Therefore, to conduct a valuation analysis of the impact of changes of the IROWC-P some research will need to be done. The impact pathway identified, phosphorus loading, is something that is not possible to quantify with the use of the proposed indicator. This gap renders the economic analysis more difficult because hypothesis and correlation will need to be established.

Actually, some research is being done in Quebec and Manitoba to correlate the results of the indicator with water quality. This is necessary for the development of a methodology to assess the economic impacts related with the changes in the indicator. This work could also help identify which factor or site characteristics influence the most the results of the indicator. Those relations will help to fully understand the mechanisms involved in the calculation of the indicator and which one has the most impact on its result.

Below are some recommendations about the work that could be done to address the gap in the transport model for the economic analysis related to the IROWC-P.

- **Improved understanding of the different components involved in the calculation of the IROWC-P**. This should lead to a better comprehension of what is happening in terms of phosphorus behaviour and movement. It also should improve our knowledge of which parameters have the most influence in terms of risk of water contamination. The new improved version of the indicator will need to be analyzed and its performance evaluated. The transport-hydrology component should allow a better understanding of what is happening at the polygon level in terms of phosphorus movement toward water and therefore, risk of water contamination. This component could be an important part of the development of the economic analysis methodology.

- **Correlation between the indicator and the water quality**. The results of the ongoing research should show the extent of the adequacy of the use of the indicator. They would also be used to adjust the weighted factors (Table 4-1) and therefore improve the calculation of the indicator.
- **Water quality improvement.** A more thorough assessment of the potential for using the probabilities of exceeding a given water quality standard to establish when an activity can be practiced, in order to assess the economic impacts.

5. Impact Pathway Analysis for Changes in Wildlife Habitat Availability

5.1 Indicator Overview

Among the most important determinants of wildlife biodiversity is the availability and quality of habitat. The conversion of natural lands to agriculture can lead to declining habitat, but it is important to also note that agriculture does offer more suitable habitat than other types of development such as urbanization (McRae et al. 2000).

The current *Availability of Wildlife Habitat on Farmland Indicator* (AWHFI) "identifies the ways in which various wildlife species use habitats in the agricultural landscape and relates this to changes in the area of these habitats." This indicator is developed using a habitat availability matrix – a chart relating habitat type found on agricultural land (cropland, summerfallow, tame or seeded pasture, natural land for pasture, and all other land) to habitat used by wildlife (birds, mammals, amphibians and reptiles). An important unit in the indicator development is the *habitat use unit* – defined as the number of ways a habitat is used by all species using the habitat. The share of habitat use units for the different habitat types is shown in Table 5-1. The indicator then looks at which habitat types in the agricultural landscape support the most wildlife use and whether these types are increasing, decreasing or remaining constant (McRae et al. 2000).

Table 5-1. Agricultural habitat types and associated habitat use units in 1996 (from McRae et al. 2000).

	Share (%) of farmland (1) and share of total habitat use units (2) associated with various agricultural land uses											
Ecozone	Total farmland area evaluated (1000 ha)	Cropland		Summerfallow		Tame or Seeded Pasture		Natural Land for Pasture		All Other Land		Total primary plus secondary habitat use units
		1	2	1	2	1	2	1	2	1	2	
Pacific Maritime	139	49	7	<1	<1	11	3	26	17	14	73	3048
Montane Cordillera	1532	16	9	<1	<1	9	3	62	17	13	70	4011
Boreal Plains	13 445	49	13	5	<1	10	3	24	14	12	69	3098
Prairies	41 853	53	17	13	<1	5	4	24	19	5	59	3865
Boreal Shield	1245	37	8	1	<1	9	3	24	14	29	75	3262
Mixedwood Plains	6294	75	11	<1	<1	6	3	10	14	9	71	3784
Atlantic Maritime	1546	40	12	<1	<1	8	3	13	12	39	73	2792

This indicator is currently being revised "to improve analytical soundness, measurability and policy relevance" (Javorek et al. 2003). In particular, the new indicator will be weighted by area through a *habitat use hectare* (HUH) measurement to increase the sensitivity to land use changes. The HUHs will "reflect the relative acreage of contributing crops or land uses (relative value of the landscape to wildlife), by multiplying the habitat use units by the amount of land in the corresponding habitat type, and adding up the results" (Javorek et al. 2003).

Additionally, the new indicator will include the number of species, the percentage of area summed across habitat types for each province. More specifically, the changes to be made will include:

- creating a database for the habitat suitability matrices;
- independent interpretation of habitats under the "All Other Land" category;
- expansion of taxa included in the matrices;
- addressing habitat quality and landscape structure issues;
- increasing indicator sensitivity to land use changes by the use of "Habitat Use Hectares";
- making the indicator interpretable at various scales (National, Provincial, EcoZone, EcoRegion, EcoDistrict); and,
- adding a species at risk component.

One of the current limitations of the indicator is the definition of the "All Other Land" category. This category groups under one heading land that is not suitable for habitat, such as lanes, greenhouses and farm buildings, together with some of the most suitable habitat such as wetlands and woodlots. Without the finer detail telling how much of each habitat is available, the use of this particular category in analysis is limited.

5.2 Impact Pathway Analysis

There are five main impact pathway constituents associated with the *Availability of Wildlife Habitat on Farmland indicator* (referred henceforth as *wildlife habitat indicator*) as depicted on Figure 5-1. These correspond to changes in the five farmland habitat categories including cropland area, summerfallow area, seeded and natural pasture area, and all other land area which includes wetlands.

Each of the habitat types supports a different bundle of ecosystem goods and services. Consequently, the potential impact pathways for each habitat area will also be unique. Table 5-2 presents an analysis of the different potential impact pathways associated with each pathway constituent. For each of the habitat types the potential impact pathways include:

1. Change in Cropland Area

- Food production function and livelihood ability

2. Change in Summerfallow Area

- Gas and climate regulation (source/sink for CO_2)
- Sediment, phosphorous and nitrogen loading

3. ***Change in Tame or Seeded Pasture***
   - Food production function and livelihood ability
   - Refugium and nursery habitat function

4. ***Change in Natural Land for Pasture***
   - Food production function and livelihood ability
   - Soil retention (limit soil erosion)
   - Refugium and nursery habitat function
   - Information functions (aesthetic information, recreation, cultural and artistic information, spiritual and historic information, science and education)

5. ***Change in All Other Land (buildings, shelterbelts, woodland types, wetlands)***
   - Gas regulation
   - Climate regulation
   - Disturbance prevention
   - Water regulation
   - Water supply
   - Soil retention
   - Soil formation
   - Nutrient regulation
   - Waste treatment
   - Pollination
   - Refugium and nursery habitat function
   - Raw materials
   - Genetic resources
   - Medicinal resources
   - Ornamental resources
   - Information functions (aesthetic information, recreation, cultural and artistic information, spiritual and historic information, science and education)

While it is true that the issue this habitat indicator is meant to address is the abundance of wildlife, the fact that available habitat is used as the surrogate for this issue opens up the scope of the potential impact pathways that are affected by a change in the indicator. For example, had the indicator been *abundance of wildlife*, then the impact pathways associated with a change in the abundance of wildlife would have been restricted to the full range of private and social costs that could be linked to wildlife (e.g., hunting and fishing, wildlife viewing tourism, food, etc.). But since the indicator is habitat, the scope broadens to include the range of ecosystem services impacted by a change in wetland area for example (e.g., disturbance prevention, climate regulation, soil retention, etc.).

There is therefore an inherent tension in where to draw the boundary for the scope of this analysis. On the one hand it might be limited to the primary purpose of the indicator (e.g., wildlife); however, on the other hand in order to stay true to the “full-cost” accounting principle, a change in habitat area should incorporate all the potential benefits and costs associated with it.

The output for this indicator is already an ecosystem state variable. Therefore, a transport model to convert a pressure indicator into an ecosystem state indicator, such as was necessary for the water erosion indicator, is not necessary for this habitat indicator.

Additionally, valuing changes in the areas of any one of the habitat types will require knowledge of the type of habitat change that occurred. This will essentially require determination of a net value based on two calculations: one that accounts for the change in value associated with a decrease in one habitat type; and the other that accounts for the change in value associated with the corresponding increase in another habitat type.

Table 5-2. Potential Impact Pathways (potential pathways shaded in grey).

	Wildlife Habitat (1996)				
Ecosystem Goods & Services	Cropland (wheat, canola, corn)	Summerfallow	Tame or Seeded Pasture	Natural Land for Pasture (grassland, sagebrush/shrubs, shrubs/woodland)	All other Land (buildings, shelterbelts, woodland types, wetlands)
Regulation Functions					
1. Gas regulation		Less summerfallow, less GHG emissions			
2. Climate regulation		Less summerfallow, less GHG emissions, less climate change			
3. Disturbance prevention					
4. Water regulation					
5. Water supply					
6. Soil retention		Less summerfallow, less erosion			
7. Soil formation					
8. Nutrient regulation					
9. Waste treatment					
10. Pollination					
11. Biological control					
Habitat Functions					
12. Refugium function	External cost/benefit		Minimal habitat use units	External cost/benefit	External cost/benefit
13. Nursery function	External cost/benefit		Minimal habitat use units	External cost/benefit	External cost/benefit
Production Functions					
14. Food	Direct producer cost/benefit				
15. Raw materials					
16. Genetic resources					
17. Medicinal resources					
18. Ornamental resources					
Information Functions					
19. Aesthetic information					
20. Recreation					
21. Cultural and artistic information					
22. Spiritual and historic information					
23. Science and education					
Human Wellbeing					
Security					
Ability to live in an environmentally clean and safe shelter					
Ability to reduce vulnerability to ecological shocks and stress					
Basice material for a good life					
Ability to access resources to earn income and gain livelihood	Direct producer cost/benefit		Direct producer cost/benefit	Direct producer cost/benefit	
Health					
Ability to be adequately nourished	Direct producer cost/benefit		Direct producer cost/benefit	Direct producer cost/benefit	
Ability to be free from avoidable diseases					
Ability to have adequate and clean drinking water					
Ability to have clean air					
Ability to have energy to keep warm and cool					
Good social relations					
Opportunity to express aesthetic and recreational values associated with ecosystems					
Opportunity to express cultural and spiritual values associed with ecosystems					
Opportunity to observe, study and learn about ecosystems					

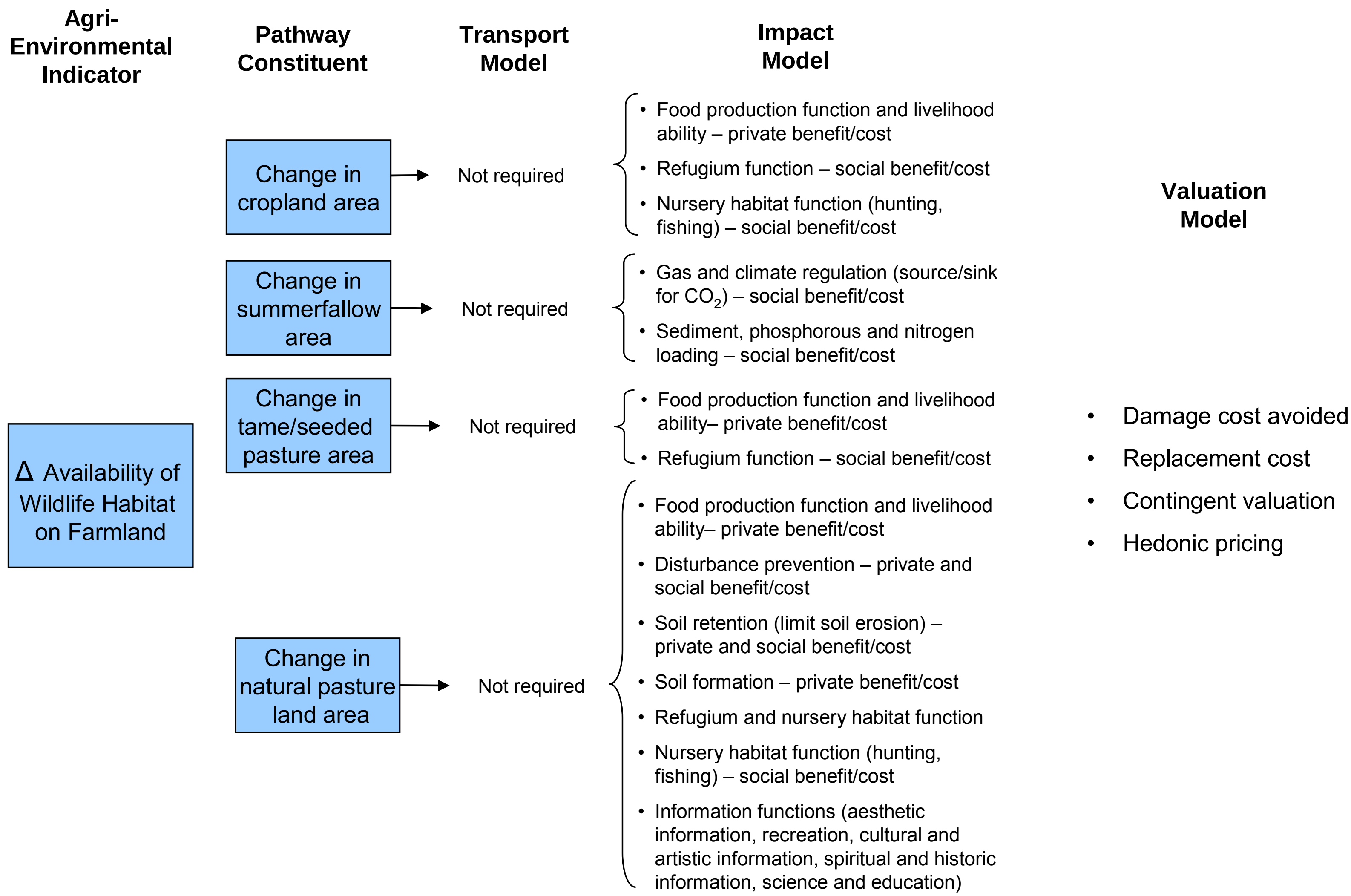

Figure 5-1. Impact pathway analysis for the Availability of Wildlife Habitat on Farmland indicator.

Pathway Constituent

Transport Model

Impact Model

Valuation Model

Change in other land area → Not required

Other land area includes buildings, shelterbelts, woodland types, wetlands. The impact pathways highlighted here are based on a change in wetland area.

- Gas regulation – social benefit/cost
- Climate regulation – social benefit/cost
- Disturbance prevention – private and social benefit/cost
- Water regulation – private and social benefit/cost
- Water supply – private and social benefit/cost
- Soil retention – private and social benefit/cost
- Soil formation – private benefit/cost
- Nutrient regulation – private and social benefit/cost
- Waste treatment- private and social benefit/cost
- Pollination – social benefit/cost
- Refugium and nursery habitat function – social benefit/cost
- Raw materials – social benefit/cost
- Genetic resources – social benefit/cost
- Medicinal resources – social benefit/cost
- Ornamental resources – social benefit/cost
- Information functions (aesthetic information, recreation, cultural and artistic information, spiritual and historic information, science and education) – social benefit/cost

- Damage cost avoided
- Replacement cost
- Contingent valuation
- Hedonic pricing

Figure 5-1 continued. Impact pathway analysis for the Availability of Wildlife Habitat on Farmland indicator.

5.2.1 Change in Cropland Area – Impact and Valuation Models

The change in level of food production associated with a change in cropland area is a relatively straightforward calculation common to agricultural economic analysis. The linkage will be a function of the type of crop and also the average expected yield. The valuation model for the change in food production can be obtained by multiplying the change in crop production by the market price for the crop. This particular impact pathway can be considered as a private cost to the producer, assuming that the change in crop production, aggregated for Canada, is not significant enough to impact market prices.

A reduction in the area of cropland will also result in a decrease of net greenhouse gas emissions. In a valuation study for the Grand River Watershed in Ontario, Olewiler (2004) said that a hectare of cropland can generate 1.92 tonnes of GHG emissions per year from fossil fuel use in Ontario[5], 0.89 tonnes per year in Prince Edward Island, and 0.938 tonnes per year in Saskatchewan and Manitoba watersheds.

Additionally, Olewiler (2004) stated that if the cropland is converted to permanent vegetative cover, approximately 1.79 tonnes of CO_2/ha/yr would be sequestered[6]. If the price of a tonne of CO_2 were $10, Olewiler notes that the incremental value of sequestration would be $17.90/ha/yr[7]. This represents an on-farm private benefit for the agriculture producer, provided it can be sold in a market, otherwise it represents a gift to the global good.

5.2.2 Change in Summerfallow Area – Impact and Valuation Models

Summerfallow habitat is found on farms primarily in the prairie (13% of farms) and boreal plains (5% of farms) ecozones. While this habitat type contains a negligible share of wildlife habitat use units relative to the other habitat types on the farm, and therefore is not of major significance to wildlife habitat, there are two impact pathways that result in off-farm social benefits/costs (external to the farm) that should be considered.

The first relates to the release of greenhouse gases to the atmosphere. Atmospheric carbon is stored in soil organic matter which is collected by plants during the growing season (SCCS 2005). The loss of organic matter that results from summerfallow practices results in carbon being released back into the atmosphere as carbon dioxide. This contributes to global climate change and there are costs associated with this that have

[5] Assuming 33% grain corn, 20% soybeans, 24% alfalfa, and 23% grains for the Grand River Watershed in Ontario
[6] Based on data from Smith, W.N. Desjardins, R.L., and B. Grant (2001). Estimated changes in soil carbon associated with agricultural practices in Canada, Canadian Journal of Soil Science 81:221-227.
[7] The current (March 2005) trading price of a tonne of CO_2 on the European market is 9.85 Euros or $15.85 CAD (Point Carbon, Available at http://www.pointcarbon.com/category.php?categoryID=745); the North American market the prices is $1.65 USD (Chicago Climate Exchange, Available at http://www.chicagoclimatex.com/)

been determined (e.g., Venema and Barg 2003). The lower the summerfallow habitat area, the lower the social cost of contributions to climate change.

The other impact pathway for a change in summerfallow areas is the corresponding increase/decrease in the amount of soil erosion. An increase in summerfallow area would correspond to an increased potential for soil erosion by water, and consequently an increase in sediment, phosphorous and nitrogen loading to water sources. The impact pathways for sediment and phosphorous loading are discussed in Sections 3 and 4, respectively.

5.2.3 Change in Pasture Area (tame and natural) – Impact and Valuation Models

The potential impact pathways associated with a change in the area of natural or seeded pasture include the following:

- Food production function and livelihood ability
- Soil retention
- Information functions (aesthetic information, recreation, cultural and artistic information, spiritual and historic information, science and education)
- Climate regulation

Food production function and livelihood ability is an on-farm cost. It is represented as a change in farm income associated with more or less pasture land. This cost can be determined by relating typical livestock production rates to the area of pasture land. Both livestock numbers and pasture area can be obtained from the Census of Agriculture.

Soil retention is both a private and a social cost. It is a private cost as it relates to more or less soil loss which can have an impact on crop productivity and income (see Section 3.3.3 for details on this impact and valuation model). It is also an external social cost as more or less soil erosion equates to more or less sediment, and phosphorous and nitrogen loading to water sources (see Section 3.3.1 and 3.3.2 for details of this impact and valuation model).

Impact on refugium and nursery habitat functions is an external social cost and refers to the maintenance of the stock of genetic diversity of wild plants and animals. Refugium is an important function for the natural pastureland that has an appreciable share of habitat use units on the farm. Valuation methods for the diversity component have been estimated using contingent valuation surveys and direct market pricing, while the nursery component typically employs only direct market pricing methods (de Groot et al. 2002). Methodologies for the valuation of this impact pathway will be covered in more detail under the "other" area category which includes wetlands. It is likely that costs of this nature will be more sensitive to changes in wetland and woodlot areas compared to changes in natural pastureland due to the significant difference in associated habitat use units.

Impacts to information functions relate to natural pasturelands providing aesthetic, recreational, cultural, artistic value, spiritual, historic information, as well as science and educational value. Methods for valuing these ecosystem functions will be presented in more detail in Section 5.3.4 under the "other" land category. Similar to the refugium and nursery functions, the valuation will be more sensitive to changes in wetland and woodlot areas compared to changes in natural pastureland.

On the climate regulation side, if the cropland or summerfallow were converted to natural pastureland, carbon could be sequestered at a rate of 1.79 tonne/ha/yr (Ontario), 1.96 (Saskatchewan/Manitoba), 0.719 (Prince Edward Island) (Olewiler 2004). Valuation of this carbon sequestration can be determined based on the current price of carbon being traded in domestic and international emissions trading (see footnote #6).

5.2.4 Change in Other Land Area (for Wetlands)

The "other land" habitat category combines land that is not suitable for wildlife habitat including lanes and farm buildings, with land that is suitable such as wetlands and woodlots. As illustrated in Table 5-1, there is a significant number of habitat use units associated with this category, due primarily to the value of wetlands for habitat. We elected to focus on the potential impact and valuation models associated with a change in the area of wetlands.

The potential impact pathways associated with a change in the area of wetlands includes the following:

- Gas regulation
- Climate regulation
- Disturbance prevention
- Water regulation
- Water supply
- Soil retention
- Soil formation
- Nutrient regulation
- Waste treatment
- Pollination
- Refugium and nursery habitat function
- Raw materials
- Genetic resources
- Medicinal resources
- Ornamental resources
- Information functions (Aesthetic information, Recreation, Cultural and artistic information, Spiritual and historic information, Science and education)

Transport Model

The transport model for a change in wetland area requires some attention for this pathway constituent. This "Other Land" component of the indicator includes several different categories including buildings, shelterbelts, woodlands and wetlands. However, the area of each of these is not provided in the indicator, nor is it available through the Census of Agriculture data. So while we present the potential impact pathways for the wetlands as an illustration of the tremendous range of ecosystem goods and services that are provided by this habitat type, valuation will not be possible until the area of wetlands can be included in the indicator.

Impact and Valuation Models

A comprehensive review of the value of natural capital in settled areas of Canada was conducted recently by Olewiler (2004). This review compiles most of the recent valuation data studies in Canada and the United States and uses that data to develop estimates for different land areas in Canada. This information, as well as other selected sources is cited in the paragraphs that follow in order to develop the impact and valuation models for wetlands and natural lands.

A comprehensive study conducted by Schuyt and Brander (2004) for the World Wildlife Fund reported on the economic value of the world's wetlands. Their study drew on the results of meta-analyses covering 89 wetland sites. The median economic value for different wetland types (covering the range of ecosystem functions associated with wetlands) are $374 USD /ha/year for unvegetated sediment, $206/ha/year for freshwater wood, $165/ha/yr for salt brackish marsh, $154/ha/year for freshwater marsh, and $120/ha/year for mangroves.

Feather et al. (1999) reported on the economic valuation of environmental benefits for the U.S. Conservation Reserve Program (CRP). This study represented a broadening of the valuation focus on soil erosion reported by Ribaudo (1989) to consider other landscape factors. Three ecosystem services in particular were chosen to demonstrate the broadened approach for targeting the CRP namely, freshwater-based recreation, wildlife viewing, and pheasant hunting. Data from this study relevant to valuing natural lands in Canada was incorporated in the valuation estimates presented in the Olewiler (2004) study; therefore, we do not cite the Feather et al. (1999) study separately. A summary of the relevant impact and valuation models reviewed in the literature are summarized in Table 5-2.

Table 5-2. Impact and Valuation Models for Natural Areas

Impact Pathway	Specific Impact	Valuation Model
Gas regulation		
Climate regulation	Carbon sequestration. GHG reduction (conversion from cropland)	(high of $26.85, best estimate of 17.90, low of 8.95)/ha/yr – Ontario[8] ($29.40, 19.60, 9.80)/ha/yr – Sask./Man. ($10.79, 7.19, 3.60)/ha/yr – PEI ($28.80, 19.20, 9.60)/ha/yr – Ontario ($14.07, 9.38, 4.69)/ha/yr – Sask./Man. ($13.35, 8.90, 4.45)/ha/yr - PEI
Disturbance prevention	Flood control	($7.50, 4.80, $2.10)/ha/yr[9] - Ontario $464 USD/ha/yr (median value for all wetland types, Schuyt and Brander 2004)
Water regulation	Water filtering	$288/ha/year (median value for all wetland types, Schuyt and Brander 200
Water supply	Water supply	$45 USD/ha/year (median value for all wetland types, Schuyt and Brande 2004)
Soil retention	Decreased soil erosion by water and impact on: - removal of sediment in water treatment - removal of phosphorous in water treat. - ditch maintenance and reservoir dredging - water reservoir dredging maintenance	 ($10.27, 5.60, 1.87)/ha/yr - Ontario[10] ($9.34, 4.62, 1.34)/ha/yr – Sask./Man. $5-500/kg P – Ontario (Olewiler 2004) ($44.50, 23.50, 2.50)/ha/yr – Ontario ($1.27, 0.69, 0.23)/ha/yr[11] - Ontario ($2.35, 1.15, 0.57)/ha/yr - PEI

[8] - Olewiler (2004) cites 1.79 tonnes/ha/yr for the Grand River Watershed in Ontario, and assumes a $10/tonne price for carbon
[9] of vegetated riparian zones consisting of 100m buffers on both sides of stream channel (Olewiler 2004).

Impact Pathway	Specific Impact	Valuation Model
	- reduced wind erosion	($4.01, 2.67, 1.34)/ha/yr – Sask./Man. (Olewiler 2004)
Soil formation		
Nutrient regulation		
Waste treatment		
Pollination		
Refugium and nursery habitat function	Hunting Wildlife viewing Habitat nursery Biodiversity	($35.04, 17.52, 8.76)/ha/yr[12] ($19.11, 10.71, 5.36)/ha/yr – Sask./Man. (Belcher et al. 2001, in Olewiler 2004) ($2.24, 1.12, 0.56)/ha/yr – PEI (Olewiler 2004) ($68.97, 34.49, 17.24)/ha/yr - Ontario[13] ($6.45, 4.16, 2.08)/ha/yr – Sask./Man. (Olewiler 2004). ($7.72, 3.86, 1.93)/ha/yr – PEI (Olewiler 2004) $201 USD/ha/yr (median value for all wetland types, Schuyt and Brander 2004) $214 USD/ha/yr (median value for all wetland types, Schuyt and Brander 2004)
Food production	Increased wildlife leading to increased damage to crops (Olewiler 2004)	($0.32, 0.64, 0.96)/ha/yr loss – Sask./Man. (Olewiler 2004) $0.64/ha – Sask./Man. (Belcher et al. 2001, in Olewiler 2004).
Raw materials	Materials	$45 USD/ha/yr (median value for all wetland types, Schuyt and Brander

[10] $9.34-$28.02/tonne of sediment (mean of $18.68) – Ontario (Olewiler 2004)
[11] $0.69/tonne sediment removed from ditches – Ontario; $2.31/tonne sediment removed from reservoirs – Ontario (Olewiler 2004)
[12] $26.68/ha/yr of improved permanent cover habitat – Ontario (Olewiler 2004)Based on pheasant hunting trips – willingness to pay study (Feather et al. (1999), in Belcher et al. (2001).
[13] $46.97/ha/yr – expenditure estimate (Ontario); $68.97/ha/yr – willingness to pay (Ontario)

Impact Pathway	Specific Impact	Valuation Model
	Fuel wood	2004) $14 USD/ha/yr (median value for all wetland types, Schuyt and Brander 2004)
Genetic resources		
Medicinal resources	Pharmaceutical research use	$20 USD/ha of threatened habitat (Simpson et al. 1996)
Ornamental resources		
Aesthetic	None-use and aesthetic	($32.04, 16.02, 8.01)/ha/yr - PEI
Recreation	Fishing Amenity recreation Water-based recreational activities attributable to conversion of agri-lands to natural areas Non-fishing recreation Hunting	($48.44, 26.42, 8.81)/ha/yr - Ontario $0.91/ha/yr – Sask./Man. (Olewiler 2004) ($33.80, 16.52, 8.15)/ha/yr – PEI (Olewiler 2004) $374 USD/ha/yr (median value for all wetland types, Schuyt and Brander 2004) $492 USD/ha/yr (median value for all wetland types, Schuyt and Brander 2004) ($2.80, 1.40, 0.70) - Ontario (Feather et al. 1999; in Olewiler 2004) ($1.37, 0.91, 0.46)/ha/yr (Olewiler 2004) $123 USD/ha/yr (median value for all wetland types, Schuyt and Brander 2004)
Cultural		
Artistic		
Spiritual		
Historic		
Science and education		

5.4 Discussion

Valuing changes in the wildlife habitat on farmland indicator presents some challenges from both methodological and data perspectives. The main challenge is that economic valuation for changes in this indicator will be most sensitive to changes in wetland and woodlot habitat areas. This area cannot be determined from the "other lands" category since farm level data on wetland and woodlots are not available in the Census of Agriculture. If changes in wetland and woodland area cannot be used to estimate both private and external costs, valuations for changes in the wildlife habitat indicator are likely to be significantly underestimated given the high percentage of habitat use units associated with these habitat areas.

Aside from this gap, the overall methodologies for valuing changes in specific impact pathways are well established. Exceptions appear to include pathways associated with many of the information type ecosystem services such as cultural, artistic, spiritual, historic, and science and education value of habitat types. Both methodology and data for these pathways appear to be lacking in the literature.

It should be noted that while review of literature showed that valuation data is available for locations across Canada (e.g., Olewiler 2004), much of this data was transferred from other sites, particularly in the United States.

There appear to be two impact pathways that potentially integrate with the valuation of other agri-environmental indicators studied in this report. One is GHG emissions from changes in the area of land under summerfallow, which should be cross-referenced with the GHG indicator to ensure that methodologies and data are shared[14]. The other is a decrease in the risk of water erosion which can occur when summerfallow land is converted to natural lands or wetlands, or an increase in water erosion if the opposite occurs.

[14] Double-counting will not be an issue if the indicators are analyzed and used individually for decision making purposes. However, double-counting can potentially be an issue when indicators are used as input into integrated economic-environmental modelling exercises.

6. Impact Pathway Analysis for Changes in GHG Emissions

6.1 Indicator Overview

Chapter 14 of “The Environmental Sustainability of Canadian Agriculture: Report of the Agri-environmental Indicator Project” (McRae et al. 2000) is called *Agricultural Greenhouse Gas Budget* (Desjardins and Riznek, 2000) – its name so chosen because the relevant indicator captures the net greenhouse gas (GHG) emissions from the entire Canadian agricultural sector.

The indicator quantifies the incremental contribution of Canadian agriculture to the risk of anthropogenic climate change. The Government of Canada, as signatory to the United Nations Framework Convention on Climate Change (UNFCCC), must report its total national GHG emissions (including agricultural GHG emissions) to the United Nations. Furthermore Canada, having ratified the Kyoto protocol (under the aegis of the UNFCCC) to reduce GHG emissions, is compelled to reduce its total emissions to combat anthropogenic climate change. Thus the agricultural GHG budget indicator is a sub-component of Canada’s national GHG accounting strategy.

Agriculture contributes approximately 10-15% of Canada’s GHG emissions (Grant et al. 2004); because the sector is intensively managed it could be a significant component of Canada’s efforts to reduce GHG emissions. For example, according to Desjardins and Riznek (2000), direct emissions from agriculture soils in Canada accounted for about 7% of agricultural GHG emissions. Although soils have lost about 25% of their carbon content since cultivated era began, recent advances in management such as no-till have reduced (and may have reversed) the loss of soil organic carbon. No-till can also have associated benefits such as reduced utilization of machinery and hence fuel consumption – the latter being a much larger overall agricultural GHG source than soils.

Agriculture is possibly the sector of the Canadian economy most at risk from the ill-effects of climate change (Wall et al. 2004). If climate change proceeds gradually, agriculture may be able to adapt, however if climate change is sudden, or if the frequency of extreme events related to climate change (such as droughts and floods) accelerates, impacts on agriculture could be drastic and may include:

- Changes in production patterns
- Increases in crop damage
- Water shortages
- New, unpredictable changes in the interactions among crops, weeds, insects and disease (Desjardins and Riznek 2000; Wall et al. 2004)

Landscape-scale GHG mitigation strategies such as no-till, which improve soil and water conservation and are important climate change adaptation responses, present an important opportunity for synergies between climate change adaptation and mitigation.

6.1.1 Indicator Units and Calculation Method

The *Agricultural Greenhouse Gas Budget indicator* is reported in units of carbon dioxide equivalents (CO_2-eq), and sums the contributions of the most important GHGs for agriculture, namely carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O).

The rationale for equating the global warming contributions from these different gasses follows a theoretical foundation provided by the Intergovernmental Panel on Climate Change (IPCC), which serves as the scientific advisory body to the UNFCCC. Research on marginal GHG damages has determined that different GHGs with different residency times in the atmosphere and with different heat-trapping properties can be compared on an equivalent basis using the concepts of Radiative Forcing (RF) and Global Warming Potential (GWP) concepts. A simple definition for RF is, "the perturbation in W/m^2 of the planetary energy balance by a climate change mechanism." The RF for a particular GHG describes its heat-trapping characteristics, which do decay over time. The rationale for introducing the RF concept is that the global mean RF can be related to the equilibrium global-mean surface temperature response, Ts, according to the following equation (Fuglestvedt et al. 2001):

$$\Delta T_s = \lambda * RF$$

Where is a climate sensitivity parameter with units K / (W/m^2).

An important underlying RF concept is that GHGs are well-mixed in the atmosphere within a short time after emission. GHGs therefore have the same RF influence regardless of the emission location, thus providing the physical rationale for the international fungibility of emissions credits under the Kyoto Protocol. The concept of Global Warming Potential is a direct extension of the RF concept and facilitates comparison to the largest (by volume) GHG, namely carbon dioxide (CO_2). GWP is defined as the time integrated commitment to radiative forcing from the instantaneous release of 1 kg of an arbitrary GHG relative to that of 1 kg of the reference gas CO_2, formally stated as follows:

$$GWP(H)_i = \frac{\int_0^H RF(t)dt}{\int_0^H RF_{CO_2}(t)dt} = \frac{AGWP_i}{AGWP_{CO_2}}$$

Where GWP(H) is the global warming potential over the time horizon, H, expressed as a ratio of the absolute global warming potential (AGWP) of the GHG of interest to that of CO_2. The UNFCCC adopted a 100 year time horizon for the purposes of the Kyoto Protocol. All GHGs emission can thus be consistently compared and inventoried according to their GWP, which has the units of CO_2-equivalents (CO_2-eq). By definition CO_2 has a GWP of 1. Table 6-1 shows the GWP for different GHGs.

Table 6-1. Global warming potential of different GHGs (CO_2-eq).

Gas	Chemical formula	GWP (100 years)
Carbon Dioxide	CO_2	1
Methane	CH_4	21
Nitrous Oxide	N_20	310
HFCs		
HFC-23	CHF_3	11700
HFC-32	CH_2F_2	650
HFC-41	CH_3F_2	150
Perfluorocarbons		
Carbon Tetrafluoride	CF_4	6500
Carbon Hexafluourice	C_2F_6	9200
Perfluoropropane	C_3F_8	7000
Sulphur Hexafluoride	SF_6	23900

For the purposes of the Agricultural Greenhouse Gas Budget Indicator only the first three gases (CO_2, CH_4, and N_2O) are deemed non-negligible and are accounted for using an agroecosystem budget framework depicted schematically in Figure 6-1.

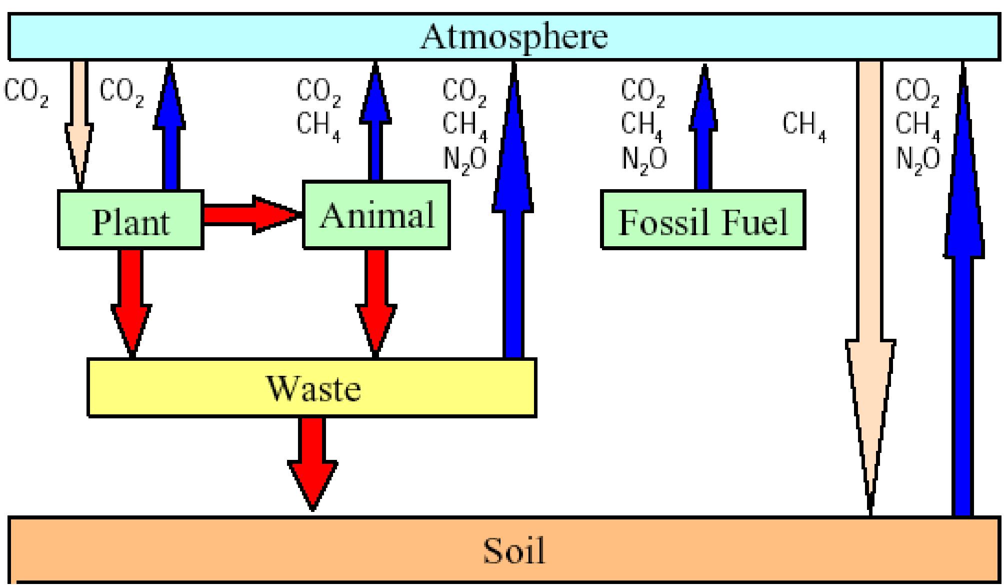

Figure 6-1. Principal sources and sinks of greenhouse gases associated with agro-ecosystems (source: Desjardins and Riznek 2000, p. 134).

Details on the accounting procedures for the three major agricultural GHG constituents are reviewed below.

Carbon Dioxide
Carbon dioxide emissions from soils were estimated using an extensively reviewed soil carbon exchange model known as CENTURY (Smith et al 2000; Smith et al 2001), which accounts for agricultural management practices, including planting, fertilizer application, tillage, grazing, and addition of organic matter. Fossil fuel use associated with agriculture management practice is also estimated using an analysis framework developed for Canada's national GHG inventory (Olsen et al 2003). The carbon dioxide emissions associated with fuel consumption and the manufacture of fertilizers and machinery, however, are attributed to the transportation and manufacturing sectors, and are considered indirect agricultural emissions. For completeness, Desjardins and Riznek (2000, p. 138) present the CO_2 emissions inventory with and without indirect emissions (shown in Table 6-2).

Table 6-2. Direct and indirect agricultural emissions of carbon dioxide

	1981	1986	1991	1996
	megatonnes of carbon dioxide			
Fossil fuels	9.5	7.7	8.1	9.5
Soils	7.7	7.3	5.1	1.8
Total Direct Emissions	**17.2**	**15.0**	**13.2**	**11.3**
Fertilizer manufacture, transport and application	4.4	5.5	5.1	6.6
Machinery manufacture and repair	4.7	4.3	3.9	3.7
Building construction	1.5	1.4	1.7	1.4
Pesticide manufacture	0.2	0.3	0.3	0.3
Electricity generation	1.8	1.9	2.1	2.4
Total Indirect Emissions	**12.6**	**13.4**	**13.1**	**14.4**
Total Agricultural Emissions*	**30**	**28**	**26**	**26**

*excluding food processing and transportation

Methane
Methane emissions from agriculture derive primarily from agricultural livestock in the form burping and flatulence, as well as the anaerobic decomposition of livestock manure. At present, Canada's livestock methane emissions for UNFCCC reporting are calculated using a standard IPCC methodology (IPCC 1997; IPCC 2000a).

Agricultural soils may act as either methane sinks or sources depending on moisture conditions, and thus the likelihood of anaerobic decomposition. Methane emissions from waterlogged areas are estimated by multiplying the total area of wet soils by an average emission factor based on measurements in Canada. Methane absorption by agricultural soils is estimated using an empirically derived value appropriate for Canada (Desjardins and Reznik 2000, p.134). Summary methane accounting values are shown in Table 6-3

Table 6-3. Agricultural Emissions of Methane

	1981	1986	1991	1996
	megatonnes of carbon dioxide equivalent			
Livestock	17.8	15.7	16.2	18.4
Manure	4.4	4.0	4.0	4.4
Soils	– 0.3	– 0.3	– 0.3	– 0.3
Total agricultural emissions	**22**	**19**	**20**	**23**

Of note is the dominance of livestock in the methane tally; the IPCC Tier-1 methodology used to arrive at this figure is a lumped parameter approach, which uses only the animal population and average emissions per animal. Boadi et al (2004, and Ominski et al (2005), have recently proposed new methodologies based on Canadian research that account for animal weight, age, gender and feeding systems, as well regional differences in animal genetics and feeding/management strategies. According to the IPCC (2000a), countries that employ a more sophisticated Tier-2 methodology with the aforementioned refinements can improve emissions estimates and reduce uncertainties.

Nitrous Oxide

Agricultural emissions of Nitrous Oxide (N_2O) occur in three major categories:

1 Direct emissions from agricultural fields
2 Direct emissions from animal production systems
3 Indirect emissions derived from nitrogen that came from agricultural systems.

The first category, direct emissions from agricultural fields includes:

- Mineral fertilizers applied to agricultural soils
- Animal manure used as fertilizer
- Nitrogen-fixing crops
- Crop residues
- The cultivation of organic soils.

The second category, direct emissions from animal production systems includes those from animal wastes (during collection and storage) and grazing animals (direct deposit onto pastures). The third category, indirect emissions, includes those associated with nitrogen fertilizer and animal manure applications such as:

- Volatilization and atmospheric deposition of ammonia and various oxides of nitrogen, and
- Nitrogen leaching and runoff.

Desjardins and Riznek (2000) used IPCC parameters to quantify N2O emissions from Canadian agriculture. Their summary data is shown in Table 6-4.

Table 6-4. Agricultural emissions of nitrous oxide
(source: Desjardins and Riznek 2000, p. 136)

	1981	**1986**	**1991**	**1996**
	megatonnes of carbon dioxide equivalent			
Fertilizers	3.5	3.5	3.4	4.8
Manure	3.3	3.0	3.2	3.5
Nitrogen-fixing crops	2.3	2.8	3.0	3.9
Crop residues	4.7	4.7	4.7	5.5
Organic soils	0.1	0.1	0.1	0.1
Total soils	**13.9**	**14.1**	**14.4**	**17.8**
Animal production systems	**6.9**	**6.2**	**6.7**	**7.6**
Total indirect emissions	**9.9**	**9.5**	**9.6**	**11.8**
Total agricultural nitrous oxide emissions	**31**	**30**	**31**	**37**

Recently researchers have applied the DeNitrification-DeComposition (DNDC) model to assess the impacts of land use management on N_2O emissions in Canada (Smith et al 2004). The DNDC model consists of four interacting submodels: thermal/hydraulic, crop growth, decomposition, and denitrification. The DNDC model had been previously calibrated with experimental data from eastern and western Canada (Smith et al 2002) and previously validated in a project that compared model results to those estimated using the IPCC methodology (Li et al 2001). Smith et al concluded from their DNDC analysis that the conversion of cultivated land to permanent grassland in Eastern Canada (where higher soil moisture levels promote denitrification) would effect the greatest reduction in N_2O emissions from agriculture. Given the increased use of, and confidence in such modeling efforts, we anticipate a trend toward the increased use of models such as DNDC for reporting agricultural GHG emissions in Canada.

6.2 Impact Pathway Analysis

6.2.1 GHG Impacts Overview

Two extremely important concepts relevant to the Agricultural Greenhouse Gas impact pathways analysis for GHG emissions are:

1 Global Warming Potential (GWP) equivalence of different GHGs (described in section 6.1), and
2 The full fungibility of GHG emissions (and emissions reductions), which underlies the Kyoto Protocol's emissions trading mechanisms.

The key idea linking these concepts is that regardless of their location and type, all GHG emissions (by virtue of near-instantaneous complete mixing in the atmosphere) are equivalent in the sense that – at the margin - they contribute equally to impacts and damages at the global scale and are thus fully fungible and tradable. **Unlike the other agricultural externalities considered in this study, which have a local or regional domain of concern, a unit of GHG emissions in Canada or anywhere else in the world (from agriculture or any other sector) is equally responsible for impacts everywhere in the world.** Thus the assessment of impacts and pathway is therefore only possible at the globally aggregated scale.

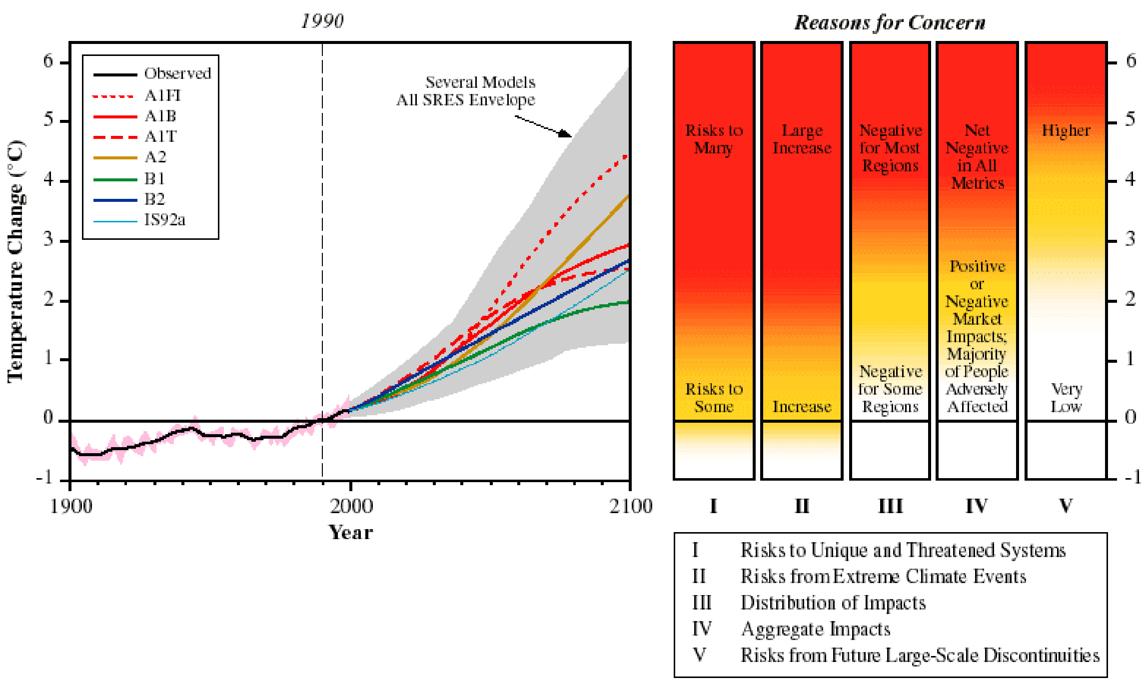

Figure 6-2. Reasons for Concern About Climate Change
(source: IPCC 2001, p. 4)

The IPCC's Third Assessment Report has conducted a relevant synthesis of global impacts from global warming, which include negative effects on health, agriculture, water supply, sea level rise, ecosystems and biodiversity. Although mediated by the global biosphere, these impacts are in effect GHG externalities. The IPCC *Impacts, Adaptation and Vulnerability Summary for Policy Makers* (IPCC 2001, p.4) attempted a graphical depiction of the magnitude of climate change and associated reasons for concern. This diagram is reproduced in Figure 6-4. Global mean annual temperature (shown on the vertical axis in Figure 6-4) is a proxy for the magnitude of climate change, however projected impacts will be a function of many factors – a partial list includes:

the magnitude and rate of global and regional changes in mean climate;
climate variability and extreme climate phenomena;
social and economic conditions; and
the capacity for adaptation.

Figure 6-4 provides a qualitative indication of the severity of risk as a function of the projected range of temperature increases. The same IPCC Summary for Policy Makers provided a useful table of probable impacts from projected increases in extreme climate events attributable to global warming, and is reproduced in Table 6-5. Despite the illustrative nature of this list, the range of impacts on human and natural systems is lengthy. Table 6-6 shows a mapping of this illustrative list of impacts to the ecosystem services and human well-being framework used throughout this study. Though illustrative, it is nonetheless straightforward to project negative impacts in all the ecosystem services and human well-being categories within this framework.

This illustrative list of impacts is largely derived from global circulation model (GCMs) results. GCMs simulate radiative forcing (RF) for global GHG emissions scenarios, and the resulting climate change. The IPCC (2000b) *Special Report on Emissions Scenarios* (SRES) provided a standardized set of global emissions scenarios for GCM-based climate change research. The SRES scenarios are based on different assumptions regarding world development pathways, north-south technology transfer, and hence expected GHG emissions. The current state-of-the-art in impact modeling is through linked socio-economic vulnerability analysis, integrated assessment and GCM modeling to GCM results for the SRES scenarios.

Figure 6-3 illustrates the generalized impact-pathway model for GHG emissions. Note that state-of-art analyses (not depicted here) uses the same development and technology transfer assumptions that drive the SRES scenarios and GCM models to infer the relative vulnerability to climate change impacts (Parry 2004a).

Table 6-5. Example Impacts Resulting From Projected Changes in Extreme Events (source: IPCC 2001, p.6)

Projected Changes during the 21st Century in Extreme Climate Phenomena and their Likelihood	**Representative Examples of Projected Impacts** (all high confidence of occurrence in some areas)
Simple Extremes	
Higher maximum temperatures; more hot days and heat waves over nearly all land areas (very likely)	Increased incidence of death and serious illness in older age groups and urban poor Increased heat stress in livestock and wildlife Shift in tourist destinations Increased risk of damage to a number of crops Increased electric cooling demand and reduced energy supply reliability
Higher (increasing) minimum temperatures; fewer cold days, frost days, and cold waves over nearly all land areas (very likely)	Decreased cold-related human morbidity and mortality Decreased risk of damage to a number of crops, and increased risk to others Extended range and activity of some pest and disease vectors
More intense precipitation events (very likely over many areas)	Reduced heating energy demand Increased flood, landslide, avalanche, and mudslide damage Increased soil erosion Increased flood runoff could increase recharge of some floodplain aquifers Increased pressure on government and private flood insurance systems and disaster relief
Complex Extremes	
Increased summer drying over most mid-latitude continental interiors and associated risk of drought (likely)	Decreased crop yields Increased damage to building foundations caused by ground shrinkage Decreased water resource quantity and quality Increased risk of forest fire
Increase in tropical cyclone peak wind intensities, mean and peak precipitation intensities (likely over some areas)	Increased risks to human life, risk of infectious disease epidemics, and many other risks Increased coastal erosion and damage to coastal buildings and infrastructure Increased damage to coastal ecosystems such as coral reefs and mangroves
Intensified droughts and floods associated with El Niño events in many different regions (likely) (see also under droughts and intense precipitation events)	Decreased agricultural and rangeland productivity in drought- and flood-prone regions Decreased hydro-power potential in drought-prone regions
Increased Asian summer monsoon precipitation variability (likely)	Increased flood and drought magnitude and damages in temperate and tropical Asia
Increased intensity of mid-latitude storms (little agreement between current models)	Increased risks to human life and health Increased property and infrastructure losses Increased damage to coastal ecosystems

Table 6-6. Likely Climate Change Impacts (illustrative) on the Ecosystem Services and Human Well-being Framework.

	Potential Impacts
Ecosystem Goods & Services	GHG Emissions
Regulation Functions	
1. Gas regulation	
2. Climate regulation	
3. Disturbance prevention	ncreased flood, landslide, avalanche, and mudslide damage
4. Water regulation	increased flood, landslide, avalanche, and mudslide damage
5. Water supply	Decreased water resource quantity and quality. Increased pressure on government and private flood insurance systems and disaster relief
6. Soil retention	Increased soil erosion
7. Soil formation	Increased soil erosion
8. Nutrient regulation	Decreased agricultural and rangeland productivity in drought- and flood-prone regions
9. Waste treatment	Decreased water resource quantity and quality
10. Pollination	Increased risk of forest fire
11. Biological control	Extended range and activity of some pest and disease vectors
Habitat Functions	
12. Refugium function	Extended range and activity of some pest and disease vectors
13. Nursery function	Increased risk of forest fire
Production Functions	
14. Food	Decreased crop yields, decreased agricultural and rangeland productivity in drought- and flood-prone regions, increased heat stress in livestock and wildlife
15. Raw materials	Increased risk of forest fire
16. Genetic resources	Increased damage to coastal ecosystems
17. Medicinal resources	Extended range and activity of some pest and disease vectors
18. Ornamental resources	Increased damage to coastal ecosystems such as coral reefs and mangroves, ncreased risk of forest fire
Information Functions	
19. Aesthetic information	Shift in tourist destinations
20. Recreation	Shift in tourist destinations
21. Cultural and artistic information	Increased damage to coastal ecosystems such as coral reefs and mangroves
22. Spiritual and historic information	Increased damage to coastal ecosystems such as coral reefs and mangroves
23. Science and education	Increased damage to coastal ecosystems such as coral reefs and mangroves
Human Wellbeing	
Security	
Ability to live in an environmentally clean and safe shelter	Increased incidence of death and serious illness in older age groups and urban poor
Ability to reduce vulnerability to ecological shocks and stress	Increased flood, landslide, avalanche, and mudslide
Basice material for a good life	
Ability to access resources to earn income and gain livelihood	Increased risks to human life, risk of infectious disease
Health	
Ability to be adequately nourished	Decreased crop yields,
Ability to be free from avoidable diseases	Increased incidence of death and serious illness in older
Ability to have adequate and clean drinking water	Decreased water resource quantity and quality
Ability to have clean air	Increased risks to human life, risk of infectious disease epidemics, and many other risks
Ability to have energy to keep warm and cool	Increased incidence of death and serious illness in older
Good social relations	
Opportunity to express aesthetic and recreational values associated with ecosystems	Shift in tourist destinations, increased damage to coastal ecosystems such as coral reefs and mangroves
Opportunity to express cultural and spiritual values associed with ecosystems	Shift in tourist destinations, increased damage to coastal ecosystems such as coral reefs and mangroves
Opportunity to observe, study and learn about ecosystems	Shift in tourist destinations, increased damage to coastal ecosystems such as coral reefs and mangroves

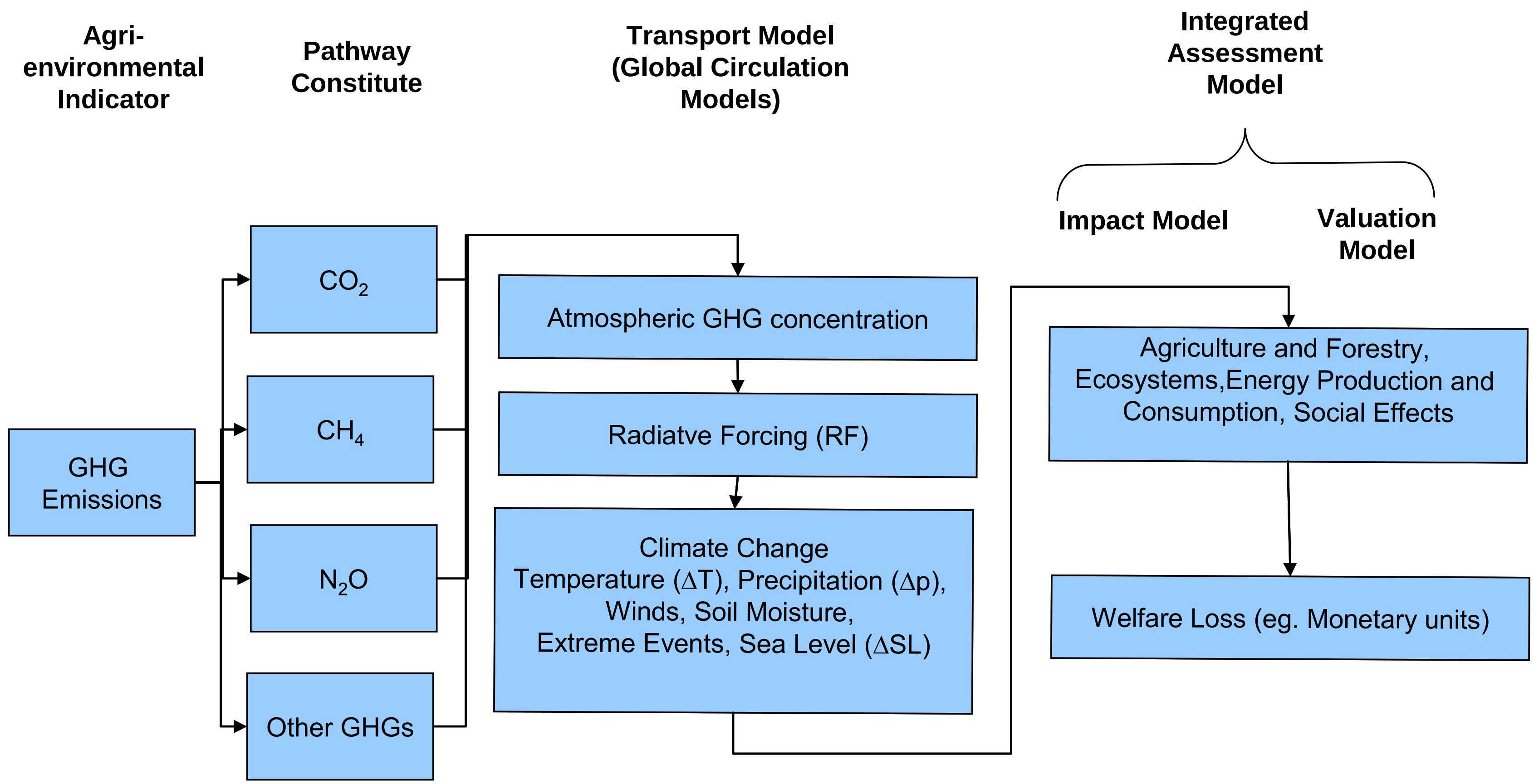

Figure 6-8. GHG impact-pathway framework

6.2.2 Impacts Evaluation using Integrated Assessment Modeling

The rigorous application of the methodological steps outlined in Figure 6-8 requires the use of global General Circulation Models (GCMs) for modelling impacts as well as multi-sector integrated assessment models (IAMs) to estimate impacts and quantify the economic outcomes resulting from climate impacts. These steps are normally outside the scope of any sector-specific analysis. The European ExternE (1999a, 1999b) project, which attempted country-specific estimates of energy production externalities, provides a useful example of sector-specific GHG externalities valuation. ExternE used two different external IAM's, FUND (climate Framework for Uncertainty, Negotiation and Distribution) developed at the Institute for Environmental Studies, Vrije Universiteit, Amsterdam (Tol 1995, 1996, 1999) and the Open Framework developed by the Environmental Change Unit, University of Oxford (Downing et al 1996). There exists no practical limitation on using these modelling results for Canadian sectoral studies (energy, agriculture, etc.) since the emitting economic sector and geographic location of GHG emissions is irrelevant to assessing the marginal impact of those emissions given the assumption of a well-mixed atmosphere. Canadian emissions are—at the margin—equally responsible for impacts everywhere in the world.

Valuating GHG externalities using IAMs follows the impact-pathways methodology (Figure 6-8), in that emissions are traced through to their impact endpoints and the damages quantified—with several important caveats (ExternE 1999a, p.1):

> the geographic location of the emissions source is irrelevant;
> the impact complexity makes a disaggregation by impact-pathway impossible; and,
> the valuation of very long-term effects introduces high uncertainties.

Both FUND and the Open Framework model the concentration of the three long-lived anthropogenic GHGs: CO_2, CH_4 and N_2O. The atmospheric GHG concentrations are used to calculate the radiative forcing, which determines average global temperature rise and sea level rise. Both models were calibrated to an earlier IPCC scenario that pre-dated the SRES scenarios (IS92a). The major impacts evaluated by both models are health, agriculture, water supply, sea level rise, ecosystems and biodiversity, and extreme events.

FUND uses a non-spatial but inter-temporal dynamic approach that incorporates sensitivity to both the level and rate of climate change. Impacts and damages are aggregated to nine world regions and derived from existing literature. The form of the damage function with respect to temperature increase is developed in considerable detail in the FUND Model. The Open Framework uses a more static approach based on first-order physical impact assessments using a GIS-based tool that links global circulation model (GCMs) results to actual physical impacts, such as loss of agricultural land and wetlands. There is much more emphasis on first-order impacts such as changes in degree-days, areas suitable for agriculture, and hydrologic balance. The Open Framework is disaggregated to the national level.

Despite the large differences in model structure between FUND and the Open Framework the marginal damage estimations for CO_2, CH_4 and N_2O remarkably varied by less than 10 per cent, 20 per cent and 40 per cent (ExternE 1999a) in the base case scenario adopted by ExternE.

The similar results between the two models are likely fortuitous; the uncertainty range is very large and important issues such as aspects of socially-contingent damages and ecosystem damages are poorly represented or not included. Human mortality is a major damage component across all impact sectors through the direct effects of extreme temperature, the spread of infectious disease, extreme weather and the socially contingent effects of resource access loss. The mortality valuation methodology is therefore a major determinant of the marginal damages estimation.

For any given assumptions on the appropriate mortality valuation method and discount rate (two of many contentious parameters), the calculated marginal damages may well represent a lower bound on actual marginal damages. The proponents of both major models acknowledge, for example, that “the impacts covered by the models used are only a fraction (of unknown size) of all climate change impacts” (Tol and Downing 2000 p.20).

6.2.3 Valuation Data and Methods

Monetizing GHG externalities: the ExternE experience

Monetizing climate change impacts requires an internally consistent valuation strategy for aggregating global damages since all GHG emissions—regardless of their physical location—are equivalent in the sense that they contribute equally to impacts and damages. Conducting research in support of ExternE, Tol and Downing (2000) tackled the global aggregation issue from four different perspectives (after Fankhauser et al. 1997):

1. from the narrow perspective of a European decision-maker concerned only EU impacts and with EU-level valuations on impacts;
2. (1), plus impacts in other regions of the world with local values;
3. (1), plus impacts in other regions with globally averaged values; and,
4. (1), plus impacts in other regions with EU values.

Although perspective 1 ignores non EU impacts, it most closely resembles the real-politic of the European decision-maker that the ExternE project was attempting to influence. Perspective 2 values damages at the expressed willingness-to-pay of people outside the EU, which the difficult and potentially objectionable implication that the value of life lost from climate change impacts in, for example Bangladesh is worth less than one lost in the EU. Perspective 3 uses globally averaged damage valuations and Perspective 4 values all impacts regardless of region at EU values. Table 6-7 lists the marginal costs per tonne of CO_2 as calculated using FUND 1.6 for three different social discount rates in year 2000 U.S. dollars.

Valuations from the FUND 1.6 model were extensively peer-reviewed for use in the ExternE project and use a statistical life valuation methodology consistent with that applied in Canadian energy sector externality studies (AMG 2000, Venema and Barg 2003). Perspective 3 was chosen as the standard central estimate assumption for ExternE work and moreover is the most philosophically consistent with the United Nations Framework Convention on Climate Change (UNFCCC) principle of "common but differentiated responsibilities" among parties (Fankhauser et al. 1997).

Table 6-7. Marginal cost of carbon dioxide emissions (USD/t CO_2).
Source: Tol and Downing (2000)

Discount Rate	EU Only	Regional Values	World Average	EU Values
0%	0.60	10.61	29.86	123.90
1%	0.46	7.12	20.13	82.55
3%	0.22	3.35	10.09	40.99

The EU-only perspective shows, not surprisingly, the lowest marginal cost, whereas modeling all global impacts at EU levels (perspective 4) has the highest costs. Table 6-7 also illustrates the influence of discount rate. Climate change is a long-term problem, hence the choice in how future impacts are discounted has critical implications for valuating the marginal costs of emissions today. Tol and Downing (2000) recap the reasons for discounting the future:

- Impatience and myopia: consumption today is preferable to consumption tomorrow.
- Economic growth: a dollar is worth more today than in the future because people in the future will be richer.
- Changing relative prices: some impacts, for example on human health, may be valued more in the future.
- Uncertainty: because consumption in the future is not certain, it is worth less than consumption today.

Of these reasons, only the third argues for a negative discount rate; however the clear recognition that climate change has multi-generational equity implications that should not be minimized leads inevitably to a much lower discount rate than used in conventional cost-benefit analysis (typically 8–12 per cent). The Canadian Analysis and Modelling Group (AMG) argued in their air quality co-benefits study that the appropriate social discount rate for their 20-year period of interest should be two to three per cent, which is consistent with general practice for regulatory analysis in the U.S. (AMG 2000).

The standardized period of interest for analyzing climate change impacts using IAMs such as FUND is 100 years. The IPCC also uses a standard 100-year period for calculating the equivalent global warming potential of different GHGs, the credits of which are then fungible under the Kyoto Protocol. These considerations argue strongly for low non-negative discounting of future climate change impacts, and therefore, consistent with the central estimates of Tol and Downing (2000), Venema and Barg (2003) adopted a one per cent social discount rate for their Canadian study, which from Table 6-7, corresponds to \$20.13 USD/t CO_2 in year 2000 dollars (\$26.36 in 1996 Canadian dollars). This estimate is in the low range of published values. As part of the IPCC Second Assessment Report, Pearce et al. (1996) surveyed the extant literature finding that the published values for CO_2 marginal costs ranged from \$5/t to \$125/t. De Leo et al. (2001) used a central estimate of 30 Euros/t and a sensitivity range of 0–250 Euros/t. Furthermore on the basis of the precautionary principle, since GHG externality estimations comprise only a fraction (of unknown size) of all climate change impacts, using a marginal cost at the extreme low end of the published range could be construed as particularly imprudent (Krewitt 2002).

Tol and Downing (2000) also provide estimates of uncertainty wherein Monte Carlo sampling was used to generate uncertainty ranges based on random sampling of the hypothesized probability distributions of the underlying parameters in the FUND 2.0 model. Tol's resulting distribution of CO_2 marginal cost estimates is approximately log-normal (skewed to the left); however some interpretation caution to this style of uncertainty analysis is advised. Firstly, Monte Carlo analysis captures only parameter uncertainty and not fundamental model uncertainty, and as such represents a lower bound on the true uncertainty. Secondly, Tol and Downing (2000) note that the probability distributions for the key parameters in FUND are not known and largely based on judgement. For the purposes of a Canadian energy sector externalities study analogous to the ExternE project, Venema and Barg (2003) re-scaled the standard deviation estimates from the FUND 2.0 model (Tol and Downing, 2000) for the World Average Valuation / one per cent discount rate case to the corresponding FUND 1.6 mean marginal CO_2 damage estimate. The adjusted standard deviation estimate is \$18.03 in 1996 Canadian dollars.

Biophysical Data and Modeling Principles for Canadian Agro-ecosystem GHG emissions

The use of standardized integrated assessment modeling results to estimate unit marginal GHG damages is a practical simplification that avoids the need for multi-sector, global integrated assessment modeling for a sector-specific issue such as monetizing Canadian agro-ecosystem GHG emissions. The relevant impact-pathway for monetization is shown in Figure 6-9, with the assumption that sectoral-level emissions models such as CENTURY, and DNDC will remain in use.

Alternatively, given the increasing emphasis on place-based approaches to valuating agricultural externalities, a distributed version of the same basic impact-pathway framework is shown in Figure 6-10, which depicts the use of *VirtualFarm* (Gibb et al, 2005). The VirtualFarm approach attempts to integrate separate findings of soil, nutrient, and livestock research work and to evaluate net greenhouse gas emissions (GHGs) from the interacting elements of current and proposed farming systems. *VirtualFarm* analyses GHG budgets from whole farming systems. The intent of the *VirtualFarm* model (now under development within AAFC) is to trace the secondary effects of individual farm practices through the whole farming system, thus providing more refined estimates of GHG emissions, and revealing the extent to which proposed mitigation practices can increase sink potential and reduce emissions throughout the system. The use of *VirtualFarm* is thus intended to avoid piecemeal policy recommendations based on individual farm system elements rather than the whole.

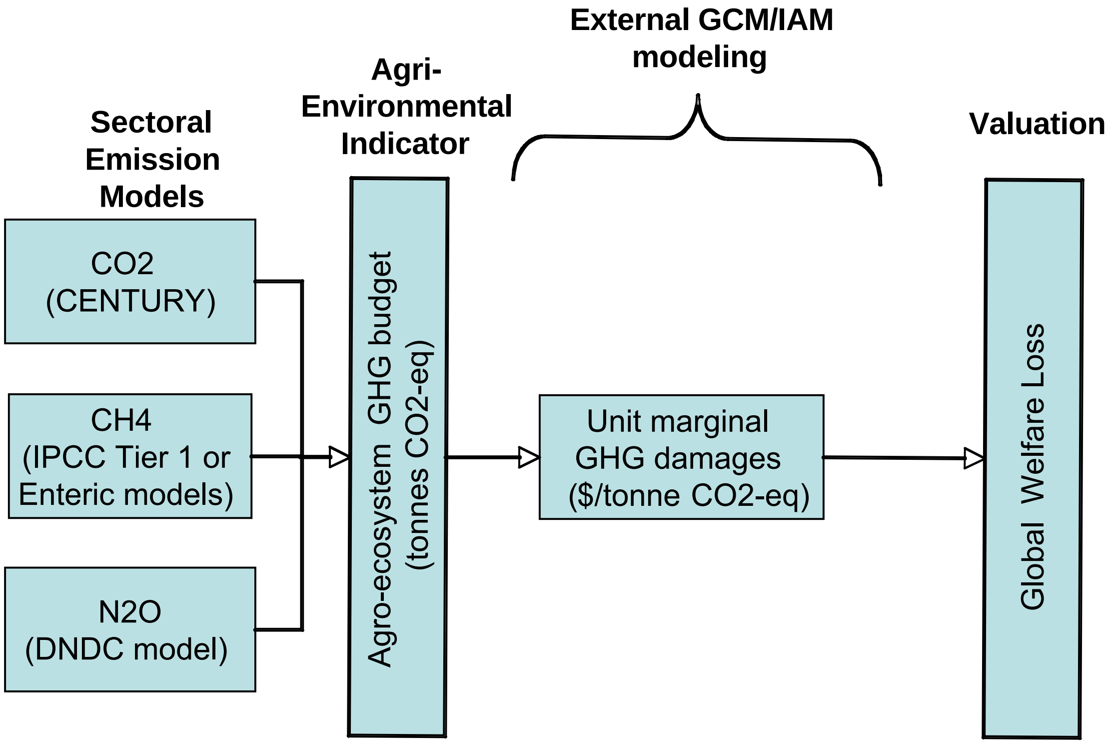

Figure 6-9. Sectoral Impact-Pathway Framework for GHG valuation

Regardless of the emission modeling approach adopted (sectoral or distributed), a major issue remains the validity of the external GCM/IAM modeling required to establish the unit marginal GHG damage value. The IPCC has explicitly avoided monetization of climate change impacts after an extremely divisive episode during the preparation of the Second Assessment Report (Parry, 2004b) that essentially pitted researchers from the north against the south over the mortality valuation principles. Human mortality is a key endpoint for a large range of climate change impacts (see Table 6-7). Standard willingness-to-pay methodology yields a lower value on human life in developing countries as lower-income people express less willingness to avoid mortality risks. The unavoidable corollary is that lives in the south are worth less than in the higher-income north - an extremely contentious outcome for the politically sensitive and very international IPCC. Tol and Downing's work (2000) is a creative, but not entirely satisfactory approach for surmounting this political issue.

IPCC syntheses have since focused on biophysical impact modeling – the relevant research continues to advance. A recent special issue of *Global Environmental Change* (Volume 14, Issue 1) reviews the application of new impact assessment approaches using models with more sectoral detail than either FUND or the Open Framework. Some of the models applied include:

Hyland (Levy et al 2004) for modeling climate impacts on natural ecosystems and terrestrial carbon stocks;
BLS (Parry et al 2004c), a global-scale general equilibrium model for the world food system as perturbed by climate change
MIASMA (van Lieshout et al 2004), a global model for the risk of malaria transmission as a function of climate change

Policy makers will inevitably demand the increased integration of these sectoral models, and the desire for some integrated monetization will in all likelihood re-emerge. It is thus incumbent on AAFC to remain abreast of these developments and periodically assess the appropriateness of resulting modeling studies for application in the Canadian context.

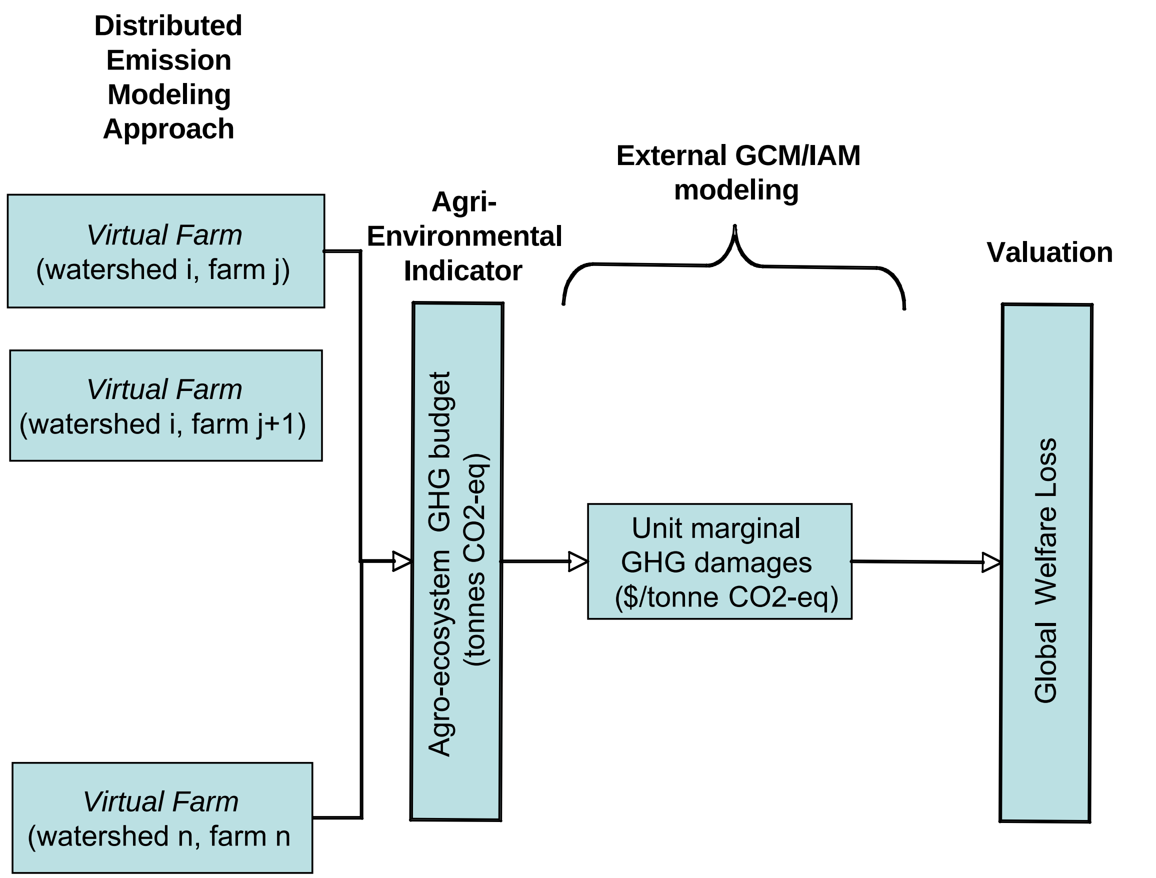

Figure 6-10. Distributed GHG impact-pathway framework for GHG valuation

6.3 Discussion

There are no methodological gaps that would prevent valuation of changes in the GHG emissions indicator. The indicator units are compatible with the impact and valuation models that have been developed for global damages resulting from climate change induced by GHG emissions. The advantage for valuation for this particular indicator in terms of data requirements is that a unit of GHG emissions in Canada or anywhere else in the world (from agriculture or any other sector) is equally responsible for impacts everywhere in the world. Therefore, credible studies completed elsewhere in the world can be considered for use in the Canadian context.

It is however, important to emphasize that the apparent simplicity of using the NAHARP indicator directly for valuation, and the logical consistency of using external global integrated assessment results, masks a very complex and contentious valuation exercise. Specifically the issue of human mortality valuation on a global scale is problematic, but fundamental to any comprehensive assessment of climate change damages. The IPCC has, since the Second Assessment Report, avoided a comprehensive global valuation of climate change damages. A non-rigorous, but much less politically problematic approach may well be to simply value agricultural GHG emissions at the international market price for emissions credits.

7. Impact Pathway Analysis for Changes in Energy Efficiency

7.1 Indicator Overview

This indicator is different from most others in the agri-environmental indicator set. The efficiency with which energy is used does not, in itself, imply any environmental impacts, so there are no direct impact pathways. Rather, the indicator is a measure of the relationship of total energy inputs to total energy outputs from agriculture, and thus (all other things being held constant), a measure of the effectiveness with which energy is used. Greater output for a given input means more efficient use. Of course, in the calculation, other things are not held constant, so the calculation gives a measure of efficiency in a given year, with all of its specifics of crop patterns, management practices, weather, and so on. The trend from year-to-year, when all of these specifics are subject to change, is more difficult to interpret.

The unit of measure used is the petajoule (PJ), which is 10^{15} (10 to the power 15) joules. A joule is the international unit for measuring energy - the energy produced by a power of one watt flowing for one second. One PJ is equivalent of approximately 0.95 billion cubic feet of natural gas, or 165,000 barrels of oil, or 280 billion kilowatt hours of electricity, or 45,000 tonnes of coal.

The data for the calculations comes from Statistics and AAFC surveys. The following details come from Piau and Korol 2004.

Liquid fuel data is collected by Statistics Canada, but it is for the entire agriculture sector, including personal and non-farm business related use. The Farm Energy Use Survey, conducted by Statistics Canada in 1981 and 1996 provides the basis for separating these out.

Fertilizer consumption data is collected by AAFC in the report "Canadian Fertilizer Consumption, Shipments and Trade", produced annually. Energy coefficients for the fertilizer products are based on studies by the Canadian Fertilizer Institute.

Buildings and machinery both embody the energy used to build them, and the energy used to maintain them. These amounts have been calculated and allocated annually based on average life spans, and included in the indicator.

Pesticide energy content is calculated from data on total pesticide costs, converted to average energy content.

Electricity usage comes from Statistics Canada

In 1992, the various inputs totalled 357.3 PJ. The inputs are distributed as shown in Table 7-1.

Table 7-1. Energy inputs to Farming, 1992. Source Piau and Korol (2004), p 7.

Input	PJ	%
Liquid fuels	152.2	43
Electricity	24.4	7
Fertilizer	74.7	21
Buildings and machinery	98.3	27
Pesticides	7.8	2
Total	357.4 PJ	100%

7.2 Impact Pathway Analysis

The indicator calculation includes energy inputs and also energy outputs from the sector. This two-part calculation allows for the exploration of some impact pathways. The inputs to the index are gasoline (motor), diesel (fuel oil), electricity, natural gas and NGLs (propane, butane, etc.), fertilizer, machinery, buildings, and pesticides (Piau and Korol 2004). Each of these inputs can be examined and its impact pathways analysed. Because of the similar impact pathways, we will combine the first three (gasoline, diesel and NGLs) into one category – liquid fuels. Figure 7-1 provides a summary of the impact pathway analysis.

The denominator of the index consists of an aggregation of the energy embodied in crops, livestock, and other farm products. Since any of the environmental impacts of the production processes will be captured by the other environmental indicators, and since the key focus of the energy use indicator is the energy itself, the embodied energy in farm products is not further dealt with here.

Since this is one of the indicators in the set that focus on inputs to agricultural activity, it brings up the issue of the life cycle impacts of energy production. The production and delivery of electricity, liquid fuels, buildings and machinery, and pesticides all involve real environmental costs. In order to ascertain the full environmental cost of the energy inputs used in agriculture, these costs must be included, despite the fact that it substantially complicates the analysis, and that all of the relevant data may not be available. The details are discussed below.

Note how the ecosystem services analysis below reveals that each of these pathway constituents, because they are at such a high level, has a number of pathway constituents associated with it. For example, for liquid fuel use there are at least four potential sub-constituents including: GHG Emissions; Air Pollution; Water Pollution; Soil Contamination. This is different than, for example, the risk of water contamination by phosphorus indicator which had only one pathway constituent, namely phosphorus (Section 4). The Energy Efficiency indicator impact pathways are more analogous to the Availability of Wildlife Habitat indicator which also had multiple pathway constituents (Section 5).

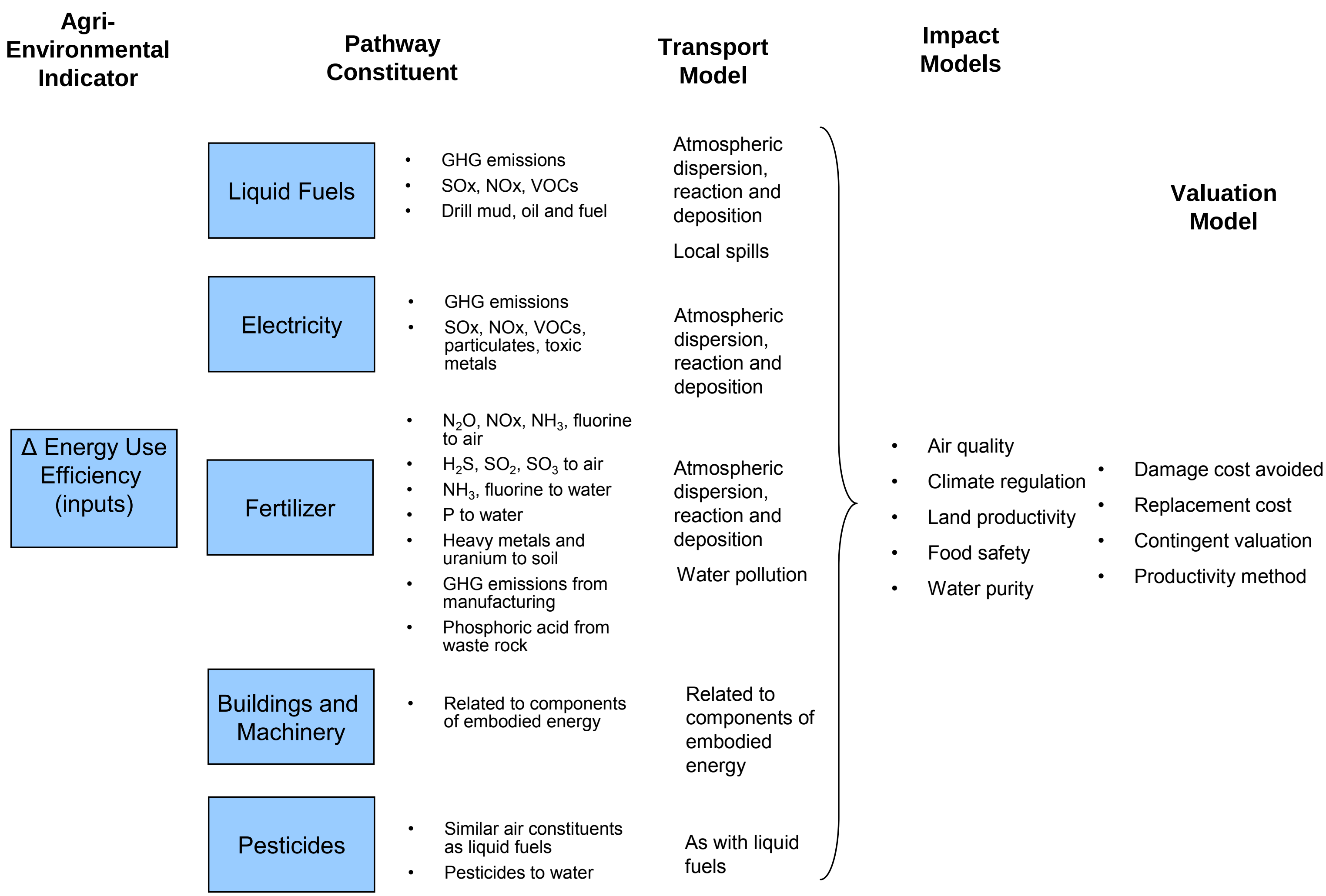

Figure 7-1. Impact of energy inputs to agriculture on ecosystem services

7.2.1 Liquid Fuels

Liquid fuels include natural gas, gasoline, diesel fuel, and natural gas liquids. These all have similar means of production (oil and gas wells, refineries), and transportation, and thus similar impact pathways. Thus they are treated together here. The use of liquid fuels will produce air pollution both in the form of GHG emissions and other air emissions. These emissions will occur both at the stage of combustion on the farm and at the earlier production and transport stages. There are both environmental and human health impacts from the emissions. In addition to air pollution, there are various possible impacts on water and soil, as summarised in the Table 7-2.

Table 7-2. Main impact pathways for liquid fuel use

<table>
<tr><th></th><th colspan="4">Impact Pathways</th></tr>
<tr><th></th><th>GHG Emissions</th><th>Air Pollution</th><th>Water Pollution</th><th>Soil Contamination</th></tr>
<tr><td>Production</td><td colspan="2" rowspan="2">There are many environmental impacts associated with activities from the upstream petroleum industry. The emissions released by the industry to the air are of concern with respect to regional air quality and greenhouse gas emissions. Air emissions from the industry include toxics, such as benzene and particulates, smog precursors, acid emissions and greenhouse gases, such as methane and carbon dioxide. Environment Canada's National Pollutant Release Inventory reports that upstream petroleum activities contribute 21% of the sulphur oxide (SOx), 13% of the nitrogen oxide (NOx), and 19% of the volatile organic compound (VOCs) in Canada. (Environment Canada 2005)</td><td>Especially at the drilling stage, but also to some degree at the production and transportation stages, water pollution may occur.</td><td>Especially at the drilling stage, but also to some degree at the production and transportation stages, soil contamination may occur.</td></tr>
<tr><td>Refining</td><td>Spills may occur that result in water pollution.</td><td>Soil contamination will occur at refinery sites – remediation will eventually be required.</td></tr>
<tr><td>Transportation</td><td>The GHG impacts were discussed in section 6 of this report.</td><td>Road transportation will cause air pollution, of the same type as in Combustion below. Pipeline or other transportation may result in small leaks.</td><td>Spills or leaks may occur.</td><td>Spills or leaks may occur. Soil contamination from spilled fuel is a widespread problem.</td></tr>
<tr><td>Combustion</td><td>The GHG impacts were discussed in section 6 of this report.</td><td>Air pollution, consisting of particulates and SOx, NOx and other chemicals, can have significant human health impacts.</td><td>Negligible.</td><td>Negligible.</td></tr>
</table>

There is a substantial literature on liquid fuels, which can be used as the basis for costing the impacts. For example, a detailed Australian study, completed in 2001, reviews 15 fuel types, and discusses health and environmental impacts (Beer et al. 2001). ExternE also has a volume on the oil and gas fuel cycles (ExternE 1995), and there is a US study that compliments the European study (ExternE US 1996). There is also good Canadian data, from the Environment Canada emissions database, are discussed in Thompson and White (2002).

The gap in the analysis of air pollution impacts occurs at the stage of the transport model. This occurs between the constituent emission stage (which is fairly well known, given the research referred to above), and the impact stage, where there is an effect on human health and the environment. As substances like NOx and SOx are emitted, they are dispersed in the atmosphere. In addition, however, they undergo chemical change, and convert into other substances. Thus the transport modelling must include both wind patterns and atmospheric chemistry calculations if it is to allow for a calculation of the impact of a given emission. For Eastern Canada, Environment Canada has such a model, called ADOM (Acid Deposition and Oxidation Model), but it is very large and expensive to run, and its geographic coverage is incomplete.

However, approximations can be made. Based on a review of over 50 site specific studies in Europe, Spadaro and Rabl (2002) derive damage costs per kilogram of emissions. Their summary table is reproduced here as Table 7-3.

Table 7-3. Damage cost per kg of pollutant emitted from power plants in Europe, from Spadaro and Rabl (2002), p 93.

Pollutant	**Impact Model**	**Valuation Model** **Euros per kg of pollutant**
PM_{10} (primary)	mortality and morbidity	15.4
SO_2 (primary)	crops, materials	0.3
SO_2 (primary)	mortality and morbidity	0.3
SO_2 (via sulphates)	mortality and morbidity	9.95
NO_2 (primary)	mortality and morbidity	small
NO_2 (via nitrates)	mortality and morbidity	15.7
NO_2 (via O_3)	crops, mortality and morbidity	1.5
VOC (via O_3)	crops, mortality and morbidity	0.9
CO (primary)	morbidity	0.002
As (primary)	cancer	171
Cd (primary)	cancer	20.9
Cr (primary)	cancer	140
Ni (primary)	cancer	2.87
Dioxins, TEQ	cancer	18,500,000
CO_2	global warming	0.029

There are a variety of assumptions that go into developing these figures, including an acceptance of the underlying modelling from which they are derived, acceptance of the dose-response calculations, and acceptance of the health valuation costs incorporated in the numbers. But this approach does hold out the promise of an approach to costing that is simpler than source-receptor modelling.

7.3.2 Electricity

The use of electricity produces almost no environmental impact, but its production and transmission has many impacts. The specifics depend on whether the electricity is generated from coal, water power, natural gas, oil, or nuclear power plants. They also depend on the location and characteristics of the transmission systems used between the generating station and the farm. The transport model issues depend on the source of the electricity. The table reproduced below summarises the results of a major study carried out be the International Energy Agency (IEA 2000a, p 8.), which compares the human health impacts of all of the main generation options.

Table 7-4. Human Health Impacts of Electricity Generation Options (IEA 2000b, p 80)

Generating system	Source of final significant impact on human health
Hydropower with reservoir	Main issue: breach of dams Risks from water borne diseases, particularly where there is irrigation
Hydropower run-of-river	Main issue: breach of dams
Diesel	Climate change Acid precipitation Photochemical smog Particulate matter
Coal	Climate change Acid precipitation Photochemical smog Particulate matter Toxic metals
Heavy oil	Climate change Acid precipitation Photochemical smog Particulate matter
Nuclear	Radioactive substances
Natural gas turbines	Climate change Acid precipitation Photochemical smog
Wind power	Negligible
Solar photovoltaic	Negligible

The most detailed analysis of the impacts of energy systems has been carried out by the ExternE project. This project was undertaken by the European Community, and produced an operational accounting framework that was subsequently disseminated, improved and applied by 50 teams from 15 European countries (Krewitt 2002). The study covers all stages of analysis, through to valuation Reports documenting the implementation of the ExternE methodology in EU member countries. The reports are available at http://externe.jrc.es/reports.html.

ExternE established a new scientific standard for quantifying power sector externalities and is being continuously updated to incorporate the latest scientific research (Spadaro and Rabl 2002). However, after exhaustive research into many different impact pathways, the conclusion was that “Without global warming, in nearly all analysed fuel cycles the mortality effects – especially due to sulphate and nitrate aerosols – dominate the results” (ExternE 1999b, p viii). In terms of the above table, the particulates and smog are the key issues. These vary greatly among different technologies and locations. However, research by IISD has quantified the human health costs of thermally generated power in Eastern Canada (Venema and Barg 2003).

That study used the available data and analytical approaches to develop estimates for the cost of externalities arising from electricity generation using coal, oil or natural gas in Eastern Canada. Two broad types of externalities were evaluated in this study—the public health costs caused by emissions of sulphur and nitrogen oxides (SOx and NOx) and volatile organic compounds (VOC) in Eastern Canada, and the marginal climate change damages caused by the emissions of greenhouse gasses (GHGs) in Eastern Canada. The data, atmospheric models and costing models that underlie this analysis are largely from Canadian federal government sources.

ExternE provided the basic conceptual framework for characterizing the emissions, dispersion, impact and cost quantification of air pollutants from the power. The IISD study adapted the emission-dispersion modelling (steps 1 and 2 of the impact-pathway approach) from earlier studies done for the federal-provincial Analysis and Modelling Group, and calculated the average SO_4 and SO_2 concentrations in individual Census divisions in Eastern Canada. This analysis was based on Environment Canada’s ADOM model. Unfortunately no comparable air pollution modelling studies exist for Western Canada and the analysis applies only to Ontario, Quebec and the Maritime provinces.

The social cost of the SO_4, SO_2 and ozone pollutants attributable to the power sector was calculated using the *Air Quality Valuation Model* (AQVM), a computer model co-developed by Environment Canada and Health Canada to estimate human health and material damage costs from air pollution within individual Census divisions. AQVM uses 1996 Canadian Census data to calculate costs within each Census division as a function of the number of exposed persons and the increase in level of concentration. Of the 17 different impact-pathways analyzed, just two accounted for almost 90 per cent of the damages, namely the mortality risk and chronic bronchitis risk from SO_4 exposure. The SO_4 mortality risk alone accounted for over 70 per cent of total damages.

The total public health externalities estimated for all SO_4, SO_2 and O_3 impact-pathways were attributed to individual fuels used for power generation on the basis of their relative emission rates of precursors to the formation of SO_4, SO_2 and O_3, namely SO_X, NO_X and VOCs. Figure 7-2 illustrates the central estimate and uncertainty bounds (one standard deviation) for the public health externalities by fuel type and reveals the high public health externality cost of coal. Two factors explain coal's high public health cost: the overwhelming dominance of SO_4 mortality risk among the various impact-pathways, and the high SOx emissions rate for coal-fired power compared to other fuels. The public health cost for gas is underestimated because it does create SOx emissions, but in upstream production stages (not accounted for in the IISD study) and not at the point of combustion.

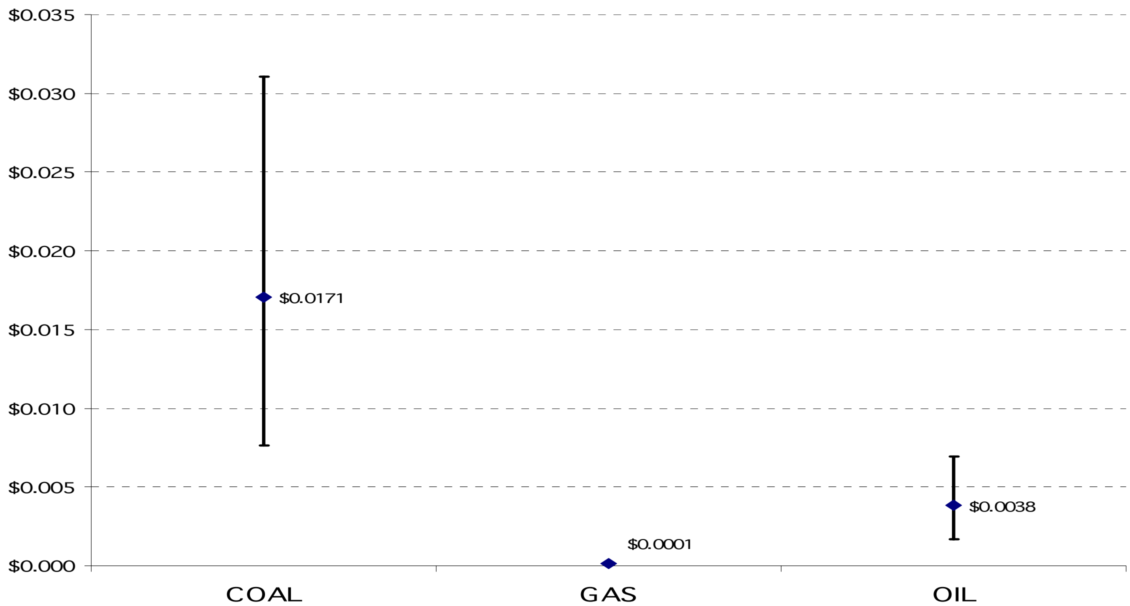

Figure 7-2. Thermal power air quality externalities in Canada ($/kWh).

Global warming damages from GHG emissions constitute the other major externality category that needs to be considered in electricity generation impact pathways. A very helpful new study published by the Commission for Environmental Cooperation (established under NAFTA) lists emissions for each generating station in Canada, as well as those in the USA and Mexico (CEC 2004). The methodology for calculating the environmental cost of a tonne of CO_2e emitted is discussed in section 7-3 of this report.

The figure below shows central estimate and uncertainty bounds in the IISD study for the aggregate air quality and global warming externalities attributable to the thermal power sector in Eastern Canada. These estimates are approximately half those of a similar ExternE study in the U.K. The differences can be explained in part by the lower population density in Eastern Canada and hence lower total exposure to air pollutants emitted by the power sector.

The results of this study can be considered a conservative first estimate as a large number of known impacts could not be evaluated because either the data, the damage function or both were not available.

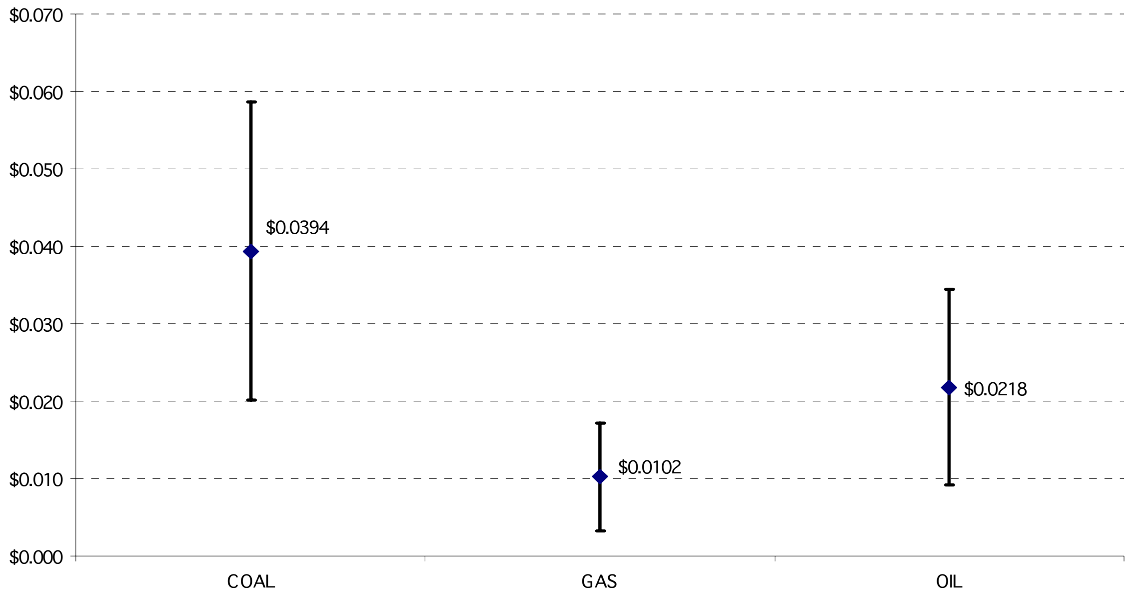

Figure 7-3. Thermal power aggregate externalities in Canada ($/kWh).

7.3.3 Fertilizer

The fertilizer industry has three main products, each with very different production processes, and all of which are produced in Canada. Nitrogen fertilizers are primarily made from natural gas, and then shipped in liquid (ammonia) or solid form (as urea, which includes phosphate as well). Phosphate fertilizers are produced from sulphuric acid and phosphate rock, and are shipped in solid or liquid form. Potash is mined and refined as a solid, and shipped in solid form. Each of these manufacturing processes and transportation systems has its own impact pathways that need to be considered. The following table outlines the key impacts at the production stage.

Table 7-5. Environmental impacts from fertilizer production (Derived from UNEP 1996)

Impact Pathway	Description
Nitrogen compounds to air	All of nitrous oxide (N_2O), ammonia (NH_3), and nitrogen oxides (NOx), are emitted to the air by natural processes, but their concentrations in the atmosphere are increased through the production and use of fertilizers. Nitrous oxide is a potent greenhouse gas, ammonia is can contribute to the formation of atmospheric particulates causing human health problems, and nitrogen oxides also contribute to particulate formation

Impact Pathway	Description
	and acid rain.
Ammonia into water	Fish can die at elevated ammonia levels in streams.
Fluorine emissions to air and water	Phosphate rock normally contains fluorine, which can be released in the manufacturing process as silicon fluoride (SiF_4) or hydrogen fluoride (HF). Both will cause damage to vegetation if concentrations are high enough, but the main problem is disposal of the liquid effluents that result from the wet scrubbers used to remove the compounds from the stack gas in the mill.
Sulphur compounds to air	Processes may release hydrogen sulphide (H_2S), sulphur dioxide (SO_2) and sulphur trioxide (SO_3), which react rapidly with water in the atmosphere to form sulphate aerosols, with significant potential human health risk, as well as environmental damage through acid rain.
Heavy metals to soil	Phosphate rock contains varying quantities of mercury, lead, and cadmium, of which the last poses the most immediate problem. When coupled with naturally occurring levels, the deposition on soils can be a problem, because cadmium is toxic and bio accumulates. Vegetable production for human consumption is a particular area of concern.
Uranium to soil	Uranium is also a constituent of phosphate rock, and some ends up in the fertilizer and in the soil.
Waste disposal	Phosphate rock treated with sulphuric acid produces phosphoric acid and phosphogypsum, which is essentially a waste material. It can occupy a large surface area and require ongoing monitoring. Potash mining also produces large volumes of waste.
GHG emissions	Especially in manufacturing ammonia from natural gas, but also in other processes using energy, there are substantial GHG emissions.

The varied transport models involved in the production of fertilizers will lead to several impact models, which are outlined above. While there is data on many of these impact models, and valuation model studies on some of them, following each pathway will be a substantial exercise.

7.3.4 Buildings and Machinery

The energy embodied in buildings and machinery is the total amount of energy used to bring the building or machine to its current state. This includes the energy used to make or gather the raw materials, processing or manufacturing them into various items, transporting them, and then manufacturing or assembling the final building or machine. The schematic below gives an indication of the commonly used approach for buildings.

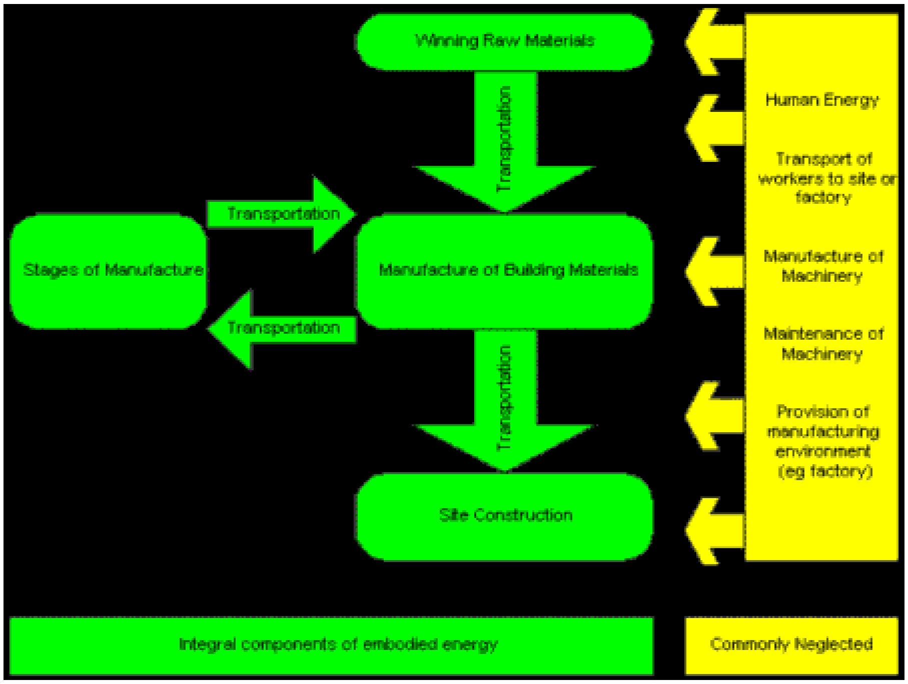

Figure 7-4. Components of embodied energy[15]

The total amount of energy obviously depends on the details of the materials used, the construction methods, etc.

A few energy coefficients are given below[16]:

Material	**Unit**	**Energy Coefficient Mj per unit**
Timber, rough	m^3	848
Timber, air-dry, treated	m^3	1,200
Timber Glulam	m^3	4,500
Timber, kiln-dry, treated	m^3	4,692
Timber, form work	m^3	283
Plywood	m^3	9,440

[15] Source: University of Brighton Faculty of Science and Engineering web page, at: http://www.brighton.ac.uk/environment/research/sustainability/elbru/embodied_energy.htm, accessed March 9, 2005
[16] Source: Home Energy Magazine Online January February 1995, accessed at: http://hem.dis.anl.gov/eehem/95/950199.html#95010912, accessed March 9, 2005

Building paper	m^2	7.5
Gypsum board	m^3	5,000
Glass	kg	31.5
Structural steel	kg	59
Aluminum	kg	145
Fiberglass batts	kg	150
Asphalt, strip shingle	m^2	280
Source: "Energy and carbon dioxide implications of building construction," by Andrew H. Buchanan and Brian G. Honey, *Energy and Buildings,* 20 1994.		

Each of these substances will have its own set of impact and valuation pathways, so the valuation analysis will be very complex and time consuming. The same issues will apply to machinery.

7.3.5 Pesticides

While pesticides have a low energy content, in proportion to other energy inputs to agriculture (2% of the total), the impact of their production can be very large, in catastrophic situations such as major leaks. However, in the normal circumstance, their production is similar to the refining of oil and gas, and they could be included with the path way analysis of the liquid fuel sector, for purposes of this paper.

7.4 Discussion

The discussion above suggests that the first challenge in quantifying the impacts of energy used in agriculture will be in defining the main impact pathways that need to be considered. Given the many sources of energy used, and the variations across Canada in providing that energy, the transport models will be numerous. Nevertheless, there has been a great deal of work on energy provision over the last 20 years, and many of the transport models are quite well understood. The difficulty is more one of a large mass of material to research, organize and present.

Much the same point can be made with respect to the impact model, with the major caveat that most of the impact research focuses on fairly densely populated areas. This has two implications: when the impacts are on human health, lightly populated areas have fewer people to be affected; and the concentration of people leads to some effects – such as smog – that are not felt in more rural areas. Thus there will be less data on the impact pathways in the agricultural areas of interest in this study.

Finally, the valuation models are also multiple and in some cases not studied.

8. Discussion of the Impact Pathway Analyses

This report presents an analytic framework for valuing changes in agri-environmental indicators that is based on methodologies cited frequently in the literature. This framework was applied to five agri-environmental indicators as a litmus test of the feasibility of valuing changes in the indicators. This test includes identification of key methodological and data gaps, potential linkages among the valuation of different agri-environmental indicators, and also to begin exploring potential linkages with NAHARP's integrated economic and environmental modelling and the social indicators component of the AAFC-IISD multi-year work agreement.

8.1 Methodological and Data Gaps and Opportunities

The uniqueness of the five agri-environmental indicators selected for analysis was evident from the impact pathway analysis. Each had unique pathway constituents leading to ecosystem impacts and to the services provided by the ecosystems, and consequently, each required their own analysis and review of available data. The risk of water contamination by phosphorus indicator was perhaps a noted exception, in that phosphorus loading was one of the pathway constituents for water erosion, in addition to being an indicator on its own. Subsequently, the detailed impact pathway analysis of phosphorus in the water erosion analysis was simply referenced to that for the phosphorus indicator.

The sections below discuss some of the key methodological and data gaps revealed in the in the impact pathway analyses.

8.1.1 Transport Model

The key transport model gaps that emerged from this study were dictated by the type of indicator being studied. The risk of water erosion and phosphorous contamination, and the GHG Emissions indicators are all pressure indicators from the perspective of the ecosystem. Aside from intrinsic and existence values, the economic costs and benefits are determined based on *impacts* to ecosystem services and human wellbeing. As illustrated on Figure 8-1, this requires that that the change in ecosystem state (e.g., water quality) be determined in some manner. The only indicator studied that was a state-of-ecosystem indicator was the availability of wildlife habitat indicator – area of habitat being the ecosystem state.

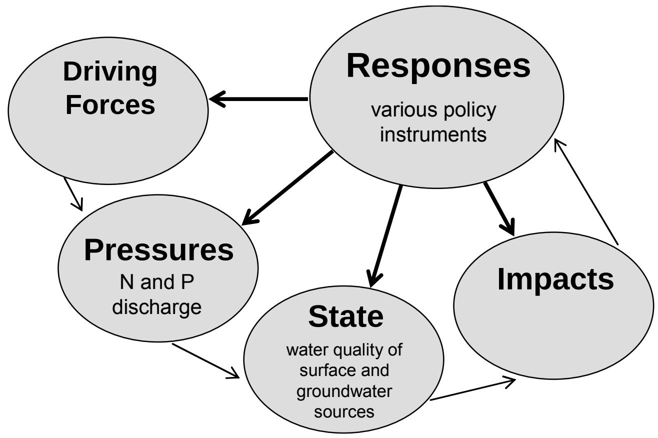

Figure 8-1. Driving forces, pressure, state, impact, response framework for indicators (framework as developed by the European Environment Agency)

This transport methodological gap exists for the sediment loading pathway of the *risk of water erosion* indicator, the phosphorous loading pathway for the *risk of phosphorous contamination*, and the *GHG emissions* indicator. Valuation is not possible for these indicator pathways without some additional transport modelling to disperse the pathway constituent to the point of ecosystem impact. For the sediment loading pathway there is a two step gap – rates of soil loss from farmland must first be related to sediment discharge to drainages and waterways, and secondly, for some impact pathways this must in turn be related to changes in water quality at the point of use (e.g., turbidity levels impacting on water treatment efforts or recreational fishing quality).

For the risk of *water contamination by phosphorous indicator*, there is a three step gap in the transport model. In addition to the two steps mentioned for the sediment loading pathway above, the phosphorous indicator is a *risk* of a pressure-type indicator, and therefore some form of model will be required to relate the risk variables of the indicator to an actual loading rate of P from farmland, or directly to changes in the quality of major waterways. As noted in the impact pathway analysis for this indicator, research is underway in Quebec and in Manitoba to establish a correlation between water quality and IROWC-P. This methodology essentially wraps the three-step gap into one relationship, and would represent savings in both data collection and analysis.

In the case of GHG emissions an accepted transport methodology exists, it is however a complex *Global Circulation and Integrated Assessment* modeling exercise – and typically external to a local or regional valuation exercise. The critical gaps are two-fold:

Developing the necessary GCM/IAM expertise, or partnering with the appropriate institutions

Resolving the political issues around mortality valuation in such global integrated assessment exercises.

The *Energy Efficiency* has several pathway constituents associated with it (e.g., liquid fuels, electricity, fertilizer, buildings and machinery, and pesticides) and even sub-pathway constituents (e.g., GHG emissions, air pollution, water pollution, and soil contamination). There are some transport model gaps associated with these pathways. For example, liquid fuel use has impact to air quality, but the emissions of the constituents of concern must be dispersed in the atmosphere before they can impact on human health in towns and cities. These issues have been tackled in other valuation studies and suitable methods are available to address such gaps in transport modelling.

Leaving the discussion on methodological and data gaps, and moving on to opportunities, the notion of the watershed as an important unit of analysis emerged from the analysis. This was the case for the *risk of water erosion* and *risk of water contamination by phosphorous* indicators – more generally, indicators related to water. As seen in the water erosion valuation work for the U.S. Conservation Reserve Program, national valuation estimates made for soil erosion were calculated for the basins of major rivers, of which there were 99 such areas in the continental U.S. Although such a national level valuation effort has not yet been conducted in Canada, researchers in Ontario have also found it necessary to use the watershed as the base unit of analysis in order to develop the loading and transport components of their valuation estimates. And in one of the studies reviewed (McRae 2000), the portions of SLC polygons situated within the watersheds were identified in order to compile the necessary data for soil erosion by water calculations.

8.1.2 Impact and Valuation Models

It is in the determination of impact and valuation of impacts that benefit transfer methods are used. This is in contrast to the transport model component which typically requires location-specific information to estimate pathway constituent loading and transport. Review of valuation data revealed that there is a relatively small world of primary valuation research and data. Many of the valuation applications found in the literature can be traced back to a limited number of unique primary data sources. This indicates two things: that benefit transfer is being used and has been accepted in the peer-reviewed literature; and there is little primary data out there.

8.2 The impact pathway model

The impact pathway model – including the pathway constituent model, transport model, impact model and valuation model – appears to be a good conceptual framework for organizing the valuation of changes in agri-environmental indicators. The impact pathway approach describes how a change in a pathway constituent (e.g., sediment loading from water erosion) causes a change in a specific ecosystem state (e.g., water quality), which in turn impacts on specific ecosystem services (e.g., water conveyance in ditches) and/or aspects of human wellbeing (ability to consume clean water). This impact can then be valued using a number of different valuation techniques.

The various stages of the Impact Pathway approach are intuitive. These stages have appeared in many different full-cost accounting exercises. For example, Ribaudo (1989) used the approach illustrated in Figure 8-2 in valuing water quality benefits from the Conservation Reserve Program in the United States. Stage 1 describes the pathway (e.g., soil and nutrient loss), stage 2 describes the dispersion of the pathway constituents into the environment (e.g., movement of soil and nutrients from field to waterways), stage 3 and 4 together describe the specific impacts (e.g., fish populations and commercial fishing), and stage 5 values the impact.

McRae (2000) used a similar approach in an environmental accounting study for stocks and flows of soil in Ontario's Grand River watershed. His approach included the following five steps:

1. Estimate levels and changes in soil stocks
2. Estimate flows of soil from cropland to water
3. Identify economic uses of water in the basin
4. Estimate damage costs to water users per unit of pollution
5. Determine aggregate value changes due to water pollution from soil flows into water.

In the context of this report, Step 1 above is analogous to changes in agri-environmental indicators determined from the indicator models. Step 2 is similar to the transport model; Step 3 is an impact model, and Steps 4 and 5 are valuation models.

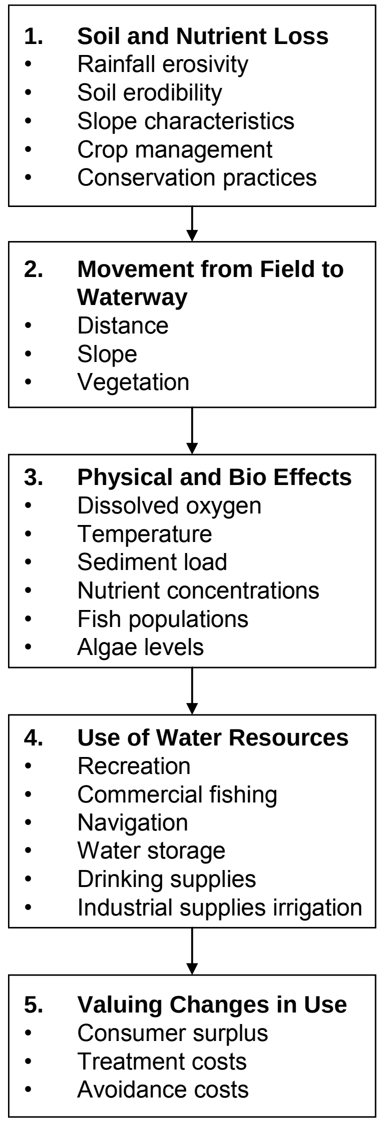

Figure 8-2. Valuation methodology used to assess water quality benefits from the Conservation Reserve Program in the United States (Ribaudo 1989).

The impact and valuation models are closely tied, and for the most part we found it convenient, as other researchers have (Ribaudo 1989), to analyse these two components together. However, for situations where the impact model is relatively complex, such as a dose-response model relating air quality concentrations of SO_2 to the occurrence of asthma, the impact and valuation models can be treated separately.

As a final note, the impact pathway analysis for an individual agri-environmental indicator, including the review of both methods and data, is a time consuming process. Some indicators have multiple pathway constituents, and each pathway constituent in turn can have multiple pathways for impact for which the methods and data are unique. The analysis for the energy efficiency indicator is a good example of the branches of analysis. Five unique pathway constituents were identified for this indicator (e.g., energy inputs including liquid fuel, fertilizer, buildings and machinery, pesticide, electricity use). In turn for example, electricity use revealed pathway constituents including GHG emissions and SO_2, each with a different set of impact pathways. The branches of this analysis were too numerous to cover in the same level of detail that was considered for indicators, such as the risk of water contamination by phosphorus which had only one pathway constituent, namely phosphorus.

8.3 Possible Linkages with NAHARP Integrated Modelling

NAHARP currently deals with two levels of integrated modelling for which potential linkages with the analyses presented in this report should be discussed. The first is the macro-level integrated economic and environmental modelling which represents one of the main components of the NAHARP program. The second is a micro-level (e.g., watershed-based) integrated modelling framework being proposed for the economic valuation component of the WEBs program.

The conceptual thinking behind the macro-level integrated economic-environmental modelling employs the Canadian Regional Agricultural Model (CRAM) which has been used as a policy tool at AAFC for many years. The unit of analysis and data collection for the CRAM model is currently based on political boundaries, whereas the unit of analysis for changes in agri-environmental indicators is more typically at the SLC polygon level. Current conceptual thinking for linking macro-level policy changes simulated by the CRAM model with changes in agri-environmental indicators is via the Land Use Allocation Model (LUAM). LUAM would allow a finer resolution for transferring demand shocks down to the farm level. For example, consider a policy situation where an ethanol manufacturing plant is forecast for a certain region. The CRAM model would project a demand shock for increased corn production to areas which already produce corn. But this may not be accurate given current land use allocation – a farm which is already producing 80% corn crops may be at capacity and not likely to take on more corn production. LUAM would allow for a more realistic picture of how the demand shock could be met[17]. With the demand shock now delivered to more of a farm or SLC polygon level, a more realistic spatial projection can then be made for changes in relevant agri-environmental indicators (e.g., changes in soil erosion). With LUAM already serving as the communication line between micro-level changes in agri-environmental indicators and macro-level policy changes, the economic valuation of changes in agri-environmental indicators could then be fed back to the CRAM in perhaps a similar manner.

So with this current conceptual picture, what are the potential challenges and opportunities for such feedback given the results of the impact pathway analysis for the five agri-environmental indicators? Essentially the longer term challenge for integrating externalities into quantitative policy analysis with a tool such as CRAM, is twofold:

[17] Personal communication with Ted Huffman, Agriculture and Agri-Food Canada

extending the LUAM logic of spatially-explicit land use allocation down to the watershed scale to drive process models that calculate externalities; and
appropriately modifying the structure of CRAM for dynamic feedback from the externality calculation routines described in the previous point.

At a more micro-level, watershed-scale integrated economic-hydrologic modelling has been proposed by Yang et al. (2004) for the Watershed Evaluation of Beneficial Management Practices (WEBs) program. This WEBs modelling proposal describes a suite of projects with the following three objectives (Yang et al. 2004):

1. Develop an integrated economic-hydrologic modelling framework that assesses the impacts of BMP adoption on pollution abatement and private (on-farm) and social benefits and costs at the farm and watershed levels.
2. Implement the modelling framework on pilot sub-watersheds.
3. Scale up the sub-watershed(s) evaluation to a regional scale (*i.e.*, river basin).

The watershed-scale integrated modelling framework uses the watershed as the base unit of analysis and proposes a watershed modelling toolbox to translate on-farm behaviour in response to policy incentives, into environmental impacts and on-farm and off-farm economic costs and benefits (see Figure 8-1).

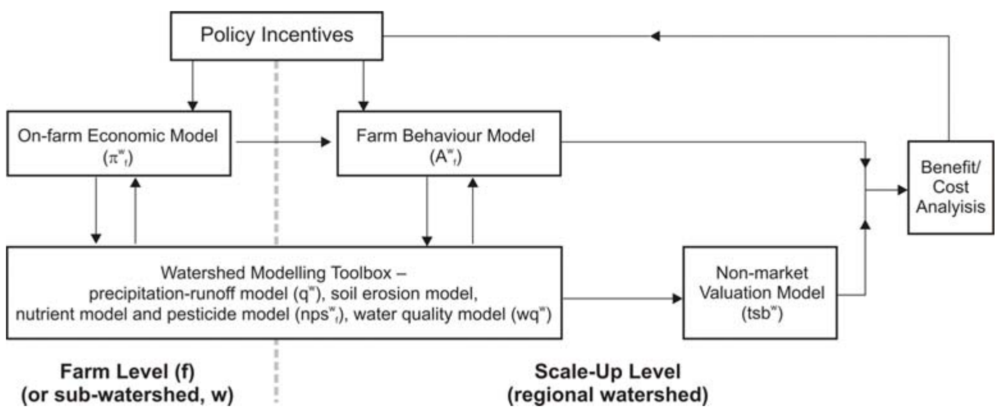

Figure 8-1. The integrated economic-hydrologic modelling framework proposed by Yang et al. (2004).

The agri-environmental indicators most directly related to Yang et al.'s proposed integrated modelling work include the risk of water erosion and the risk of water contamination by phosphorous and nitrogen indicators (and the proposed pesticide and pathogens indicators). The key aspect to discuss in relation to the analyses reported in this study would appear to be consistency in how loadings of sediment, phosphorous and nitrogen are determined for the current agri-environmental indicators and the watershed modelling toolbox proposed by Yang et al. (2004). The approaches used for market and non-market valuation of environmental impacts do not appear to vary too much based on review of the literature; therefore, consistency in valuation methodologies for individual agri-environmental indicators as studied in this report, and valuation exercises being proposed by Yang et al. (2004) would not likely be an issue.

A key issue related to the Yang et al. proposal is the feasibility of scaling up to a national level based on detailed computer-based hydrologic modelling of individual watersheds. McRae (2000) in valuing changes in soil erosion for the Grand River Basin in Ontario did not use a computer-based watershed hydrologic modelling approach (e.g., the GAMES model as used by Fox and Dickson (1990) – it was deemed impractical for his study "due to the data requirements for so large a study area." While data requirements and computational efforts are definitely considerations in scaling up, it is also worth noting that the policy environment related to watershed governance is also changing. Many provincial water strategies (e.g., Alberta and Manitoba) are requiring the development of integrated watershed management plans. Such plans are likely, although not initially, to begin using watershed-based hydrologic modelling as a tool to assist with strategy development, planning, and evaluation. So it is not inconceivable to envision in the near future, the application of watershed-based hydrologic modelling in a significant number of watersheds across the country. Therefore, while now the data requirements might seem onerous, such detailed modelling could be feasible in the not too distant future.

One aspect on which it would be beneficial to maintain consistency is in the use of consistent ecosystem goods and services and human wellbeing framework for identifying potential impact and valuation pathways. Such consistency would be beneficial from a simple terminology perspective, but also from the perspective of understanding and appreciating potential linkages with AAFC's work on social indicators. The latter point is discussed in more detail in the section that follows.

8.4 Possible Linkages with AAFC-IISD Social Indicators Work

The ecosystem services and human wellbeing framework used in this study to identify impact pathways can provide a useful mechanism for understanding conceptual linkages with the social indicators work. This linkage is particularly evident given that the framework being proposed for the AAFC-IISD project on social indicators for the agriculture sector (Figure 8-1) is also based on aspects of human wellbeing.

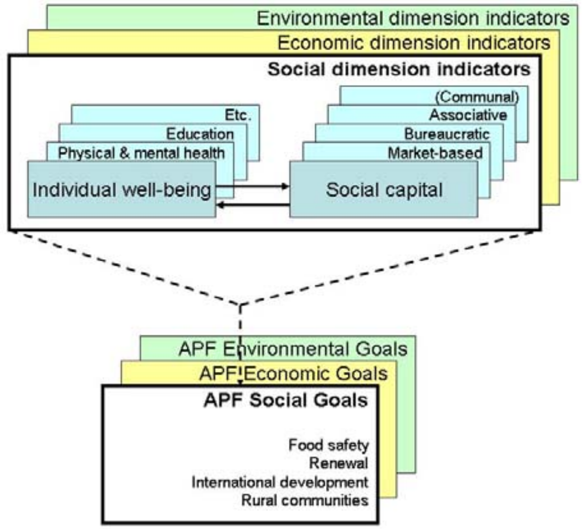

Figure 8-1. Proposed indicator framework for the AAFC-IISD project on social indicators for the agriculture sector.

The results of the Millennium Ecosystem Assessment (MA 2003) articulated the potential linkages between changes in ecosystem services and human wellbeing (reviewed in IISD 2004). Our impact pathway analysis in this study used the ecosystem services terminology described by de Groot et al. (2002) along with the MA's human wellbeing framework to help identify potential impact pathways associated with changes in agri-environmental indicators (refer back to Table 2-1 and 2-2 for details).

In Table 8-1 we highlight those linkages between ecosystem services and the aspects of human wellbeing outline that are most intuitive. Table 8-1 reveals that there are many linkages and therefore, the use of an ecosystem services framework for valuation exercises provides a means for understanding conceptual linkages with social indicators. From a valuation data availability perspective, aside from the aesthetics and recreation services, valuation data related to impacts on the information services including cultural, spiritual, historic, and science and education information services are not typically cited. Data related to impacts on human health, security (e.g., disturbance prevention via drainage) and basic material for a good life (e.g., livelihoods), are more readily available in the literature.

Table 8-1. Ecosystem functions/services – Intuitive links with human wellbeing highlighted. (based on De Groot et al. 2002)

Ecosystem Functions/Services	**Examples**	**Linkage to Aspects of Human Wellbeing (see Table 2-1)**
Regulation Functions		
1. Gas regulation	1.1 UVb-protection by O_3 (preventing disease). 1.2 Maintenance of (good) air quality. 1.3 Influence on climate (see also function 2.)	Human Health
2. Climate regulation	Maintenance of a favourable climate (temp., precipitation, etc) for, for example, human habitation, health, cultivation	Human Health, Basic material for a good life
3. Disturbance prevention	3.1 Storm protection (e.g. by coral reefs). 3.2 Flood prevention (e.g. by wetlands and forests)	Human Health, Security
4. Water regulation	4.1 Drainage and natural irrigation. 4.2 Medium for transport	Security
5. Water supply	Provision of water for consumptive use (e.g. drinking, irrigation and industrial use)	Human health
6. Soil retention	6.1 Maintenance of arable land. 6.2 Prevention of damage from erosion/siltation	Basic material for a good life
7. Soil formation	7.1 Maintenance of productivity on arable land. 7.2 Maintenance of natural productive soils	Basic material for a good life
8. Nutrient regulation	Maintenance of healthy soils and productive ecosystems	
9. Waste treatment	9.1 Pollution control/detoxification. 9.2 Filtering of dust particles. 9.3 Abatement of noise pollution	Human health
10. Pollination	10.1 Pollination of wild plant species. 10.2 Pollination of crops	
11. Biological control	11.1 Control of pests and diseases. 11.2 Reduction of herbivory (crop damage)	Basic material for a good life
Habitat Functions	***Maintenance of biological & genetic diversity (and thus the basis for most other functions)***	
12. Refugium function	Maintenance of commercially harvested species	Basic material for a good life
13. Nursery function	13.1 Hunting, gathering of fish, game, fruits, etc.	Basic material for a good life, Good social relations
Production Functions		
14. Food	7.2 Building & Manufacturing (e.g. lumber, skins) 14.2 Fuel and energy (e.g. fuel wood, organic matter) 14.3 Fodder and fertilizer (e.g krill, leaves. Litter)	Basic material for a good life
15. Raw materials	15.1 Improve crop resistance to pathogens & pests. 15.2 Other applications (e.g. health care)	
16. Genetic resources	16.1 Drugs and pharmaceuticals. 16.2 Chemical models & tools. 16.3 Test- and essay organisms	Human Health
17. Medicinal resources 18. Ornamental resources	Resources for fashion, handicraft, jewellery, pets, worship, decoration & souvenirs (e.g. furs, feathers, ivory, orchids, butterflies, aquarium fish, shells, etc.)	Good social relations Good social relations

Ecosystem Functions/Services	**Examples**	**Linkage to Aspects of Human Wellbeing (see Table 2-1)**
Information Functions		
19. Aesthetic information	Enjoyment of scenery (scenic roads, housing, etc.)	Good social relations
20. Recreation	Travel to natural ecosystems for eco-tourism, outdoor sports, etc.	Good social relations
21. Cultural and artistic information	Use of nature as motive in books, film, painting, folklore, national symbols, architect, advertising, etc.	Good social relations
22. Spiritual and historic information	Use of nature for religious or historic purposes (i.e. heritage value of natural ecosystems and features)	Good social relations
23. Science and education	Use of natural systems for school excursions, etc. Use of nature for scientific research	Good social relations

9. Conclusions and Next Steps

9.1 Key Points from the Impact Pathway Analyses

The key points emerging from the impact pathway analyses are summarized below.

- **Impact Pathway Analysis Approach**
  The impact pathway analysis conducted for each of the five focus agri-environmental indicators has proven an effective approach for identifying the important conceptual features for valuing changes in agri-environmental indicators. The use of an ecosystem services and human wellbeing framework has also proven robust for helping to identify detailed impact pathways. While many of the impact pathways were intuitive and could have been identified without using an ecosystem and human wellbeing framework, the framework was helpful in the brainstorming process.

- ***Transport Modelling***
  Of the five agri-environmental indicators studied, the output from three of the indicators can be used directly in impact and valuation modelling. These include the *Availability of Wildlife Habitat*, *GHG Emissions*, and *Energy Efficiency* indicators. The output from the *Risk of Water Erosion* and the *Risk of Water Contamination by Phosphorus* indicators will require additional transport modelling in order to assess the change in the state of the ecosystem necessary for determining impacts to ecosystem goods and services. Empirical and physically-based methods have been developed for such modelling, and it will be a matter of determining which methods are feasible given the need to replicate the analysis and aggregate to a national scale.

- ***Impact and Valuation Modelling***
  The methods for impact and valuation modelling are relatively well established for most of the impact pathways analysed in this report. Data are also available from studies conducted in Canada, the United States and internationally to allow a first level valuation. Benefit transfer appears quite common in Canadian valuation exercises, particularly related to water erosion and habitat changes. Many of the valuation exercises cited in the literature can be traced back to a relatively small subset of primary data sources. It is also the case that many of the assumptions upon which the primary data sources are not carried forward in the benefit transfer process, making it difficult to assess the credibility of the valuation.

  The watershed or water basin spatial units can potentially serve as an important unit of analysis for valuing changes in agri-environmental indicators. This unit of analysis was used in the literature for valuing changes in two of the five indicators studied – namely the water related indicators (e.g., the risk of water erosion and the risk of water contamination by phosphorus indicators). Given that such a unit of analysis would likely be required for all indicators related to soil and water quality, it may prove convenient to do so for the other indicators, but this remains to be tested.

- ***Linkages to Integrated Economic-Environmental Modelling***
  For macro-level integrated modelling, the Land Use Allocation Model (LUAM) appears to be a promising channel whereby policy changes simulated via the Canadian Regional Agricultural Model (CRAM) model can communicate with changes in agri-environmental indicators at the SLC polygon or watershed/basin level.

- ***Linkages to Social Indicators***
  The use of an ecosystem services framework for identifying impact pathways can be a useful mechanism for identifying important linkages and feedback loops between changes in agri-environmental indicators and AAFC's social indicators relating to human wellbeing.

Key findings related to each of the five agri-environmental indicators studied include the following:

- ***Risk of Water Erosion.***
  - Valuing changes in this indicator will require development of a transport model component to convert soil loss from farmland to changes in sediment and turbidity in channels and major waterways.
  - The pathway constituents for this indicator include changes in phosphorus and nitrogen loading to water, in addition to sediment loading and soil productivity loss. Therefore, valuing changes in the *water erosion* indicator need to incorporate the results from the valuations in the *risk of water contamination by phosphorus and nitrogen indicators*, and potentially the new water quality indicators being developed.

- ***Risk of Water Contamination by Phosphorus***
  - The existing indicator measures a risk of pollution, and does not contain an evaluation of actual phosphorous loading to water, which is necessary for an estimate of ecosystem impact and the associated costs and benefits. Research is underway to make the links, but it is not yet complete.
  - A range of methods are and can be considered to close this gap – among them are relating the risk indicator directly to water quality in waterways, or determining loading rates based on the indicator and transporting this load to water ways using empirically or physically-based models.

- ***Availability of Wildlife Habitat on Farmland***
  - The most influential farmland habitat types impacting on wildlife are wetlands and woodlots. The impact pathways associated with changes in these habitat areas are numerous and there is appreciable amount of valuation data available that could be used for benefit transfer. However,

changes in these habitat areas cannot be determined using the current indicator (a factor of the availability of data in the Census of Agriculture). This data gap will have to be overcome before realistic valuation estimates can be made for this indicator.

- ***GHG Emissions***
  - While there are no methodological or data gaps that would prevent valuation of changes in the GHG emissions indicator, the issue of human mortality valuation on a global scale is problematic. The Intergovernmental Panel on Climate Change is currently avoiding a comprehensive global valuation of climate change damages for this reason.
  - A non-rigorous, but much less politically problematic approach may well be to simply value agricultural GHG emissions at the international market price for emissions credits.

- ***Energy Efficiency***
  - We examined the impacts of agriculture on ecosystems caused by the impacts of the energy used in agriculture, at all stages before the energy arrives at the farm. This is the only indicator studied that dealt with inputs. A life cycle approach implies that a great many impact pathways are involved and ultimately costs must be developed for all of the important ones. This is a complex and time-consuming analysis.
  - However, a great deal of work has already been done on many of the impact pathways that will need to be evaluated, so while there would doubtless be important data gaps, policy relevant costing information can likely be found.

9.2 The Big Picture

Our understanding of the big picture for national-level valuation of changes in agri-environmental indicators is illustrated on Figure 9-1. It begins the development and testing of a methodology to analyze the impact pathways for individual agri-environmental indictors and to understand methodological and data needs and gaps. This indicator-level analysis has been undertaken for five indictors and the results presented in this report. Impact pathway analyses for the remaining agri-environmental indicators, as depicted on the right side of the illustration, represents work that remains at this level.

A possible next phase in the big picture is the actual valuation of changes in one of the five focus agri-environmental indicators at a specific spatial scale. This would be a ground-truthing stage designed to value the changes in the selected indicator for the spatial area of interest. This area could be any spatial unit. However, results of the impact analyses suggest that for the soil and water quality indicators, the watershed or water basin unit is required for determination of impacts to ecosystem services (primarily due to impacts on waterways).

The output of this analysis would be an actual dollar value estimate for the on-farm (internal or private) and off-farm (external or social) benefits and costs associated with changes in the selected agri-environmental indicator. But this calculation is an iterative process closely related to: 1. the methods and feasibility of replicating the valuation for all watershed or basins in a province; and 2. the ability to provide feedback into macro-level integrated economic-environmental modelling such as is carried out via CRAM and LUAM simulations.

Using the conceptual framework for aggregating to a provincial level, a next phase could include an analysis of the feasibility of an actual provincial-level valuation for a change in the selected agri-environmental indicator for a province. Acknowledging that data availability is likely to differ among provinces, this exercise would be an iterative process closely related to the methods and feasibility for aggregating to the national level, and the ability to provide feedback into integrated economic-environmental modelling efforts. In each basin-level, provincial-level and national-level analysis, the exercises conducted for the selected agri-environmental indicator would need to be repeated for each of the remaining agri-environmental indicators.

A national-level analysis is one of the goals of AAFC's valuation related efforts – to value the change in an agri-environmental indicator at the national level. Figure 9-1 also highlights that each level of analysis (e.g., basin, provincial, and national) provides information for different decision making processes and decision makers.

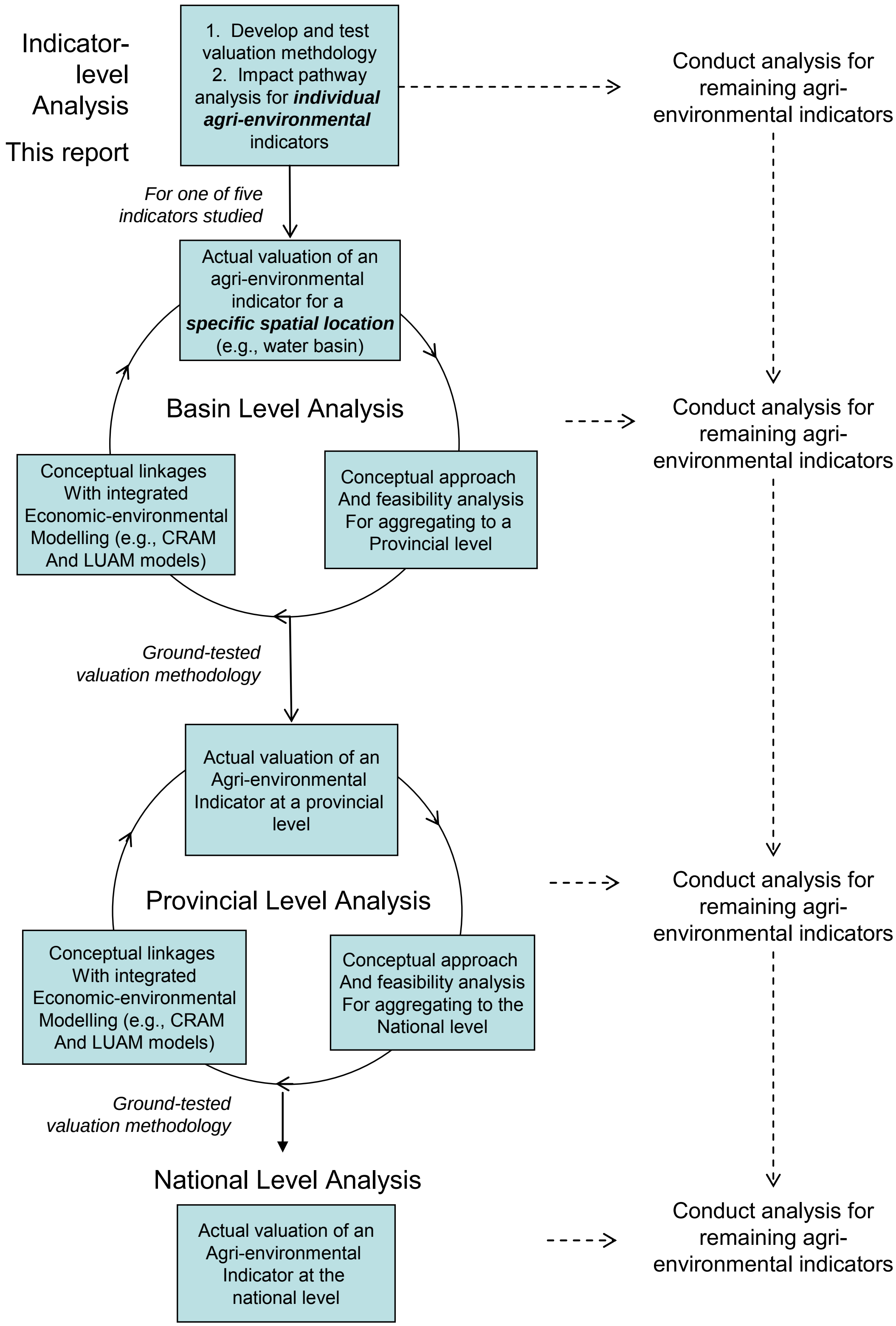

Figure 9-1. IISD's conceptualization of a process for advancing towards the valuation of changes in agri-environmental indicators at the national level.

9.2 Possible Next Steps for IISD Research

Just as the valuation literature review from year 1of this project provided the knowledge base for developing the work plan upon which this year 2 report is based, the results from this report are meant to inform the work plan for year 3 of the initiative. Based on our understanding of the *big picture*, a possible set of next steps for year 3 of the IISD-AAFC work plan on full-cost accounting could focus on a basin-level analysis in Figure 9-1. This would include the following tasks:

- Actual valuation calculation for one agri-environmental indicator for a specific watershed basin;
- Development of a conceptual framework for aggregating to the provincial level; and,
- Studying the linkages with macro-level integrated economic-environmental modelling (e.g., CRAM and LUAM).

The results of the impact pathway analyses collectively revealed that valuation of changes in agri-environmental indicators is feasible from a methodological and data availability perspective. The conceptual models for transport, impact and valuation are sufficiently articulated at this point to enable field scale application. For the 2005-06 work plan we suggest an actual calculation of one selected agri-environmental indicator for one specific watershed basin in order to develop the detailed calculation methodologies and valuation data requirements for different components of the impact pathway approach.

A base unit of analysis would need to be selected for such an application. Based on the results in this report we suggest that a specific watershed basin be identified to develop the calculations and data for transport, impact and valuation modeling. We suggest that the work plan include development of an aggregation methodology all the way up to the spatial scale of the province (e.g., assuming that all other watersheds and basins in a province have the same data available as the test watershed and basin). This work would include a feasibility assessment for aggregating the valuation from the watershed basin level to the provincial level, recognizing of course that this would be an iterative process with the development of a methodological framework for the aggregation.

The results of this suggested work would position AAFC to conduct a provincial scale valuation for a change in the selected agri-environmental indicator and to test the receptivity of decision makers in using environmental valuation information in the decision making process.

Such work would require the selection of one agri-environmental indicator and a site for the ground testing. It would seem logical that the selected indicator be one of the five agri-environmental indicators studied in this report.[18] Given this pool of indicators, there is a variety of possibilities for a ground test location:

[18] The risk of water contamination by nitrogen indicator could also be included in this list given the similarities in impact pathways to the water erosion and phosphorous indicators.

- ***GHG Emissions:*** AAFC currently has a test site in eastern Ontario (between Montreal and Ottawa) where aerial measurements of methane and NOx are taken over large areas. This could be a good site to test the GHG indicator valuation.
- ***Wildlife habitat:*** There is a good test site in the Lancaster townships in eastern Ontario.
- ***Energy efficiency:*** The Agriculture Research stations that are doing work on no till practices would likely have information on energy use efficiencies at the farm level and perhaps some costs.
- ***Risk of Water Erosion and P-Contamination***: There are currently twelve test sites that are part of the Watershed Evaluation of Best Management Practices (WEBS) project.

The WEBs sites shown on Figure 9-2 and listed in Table 9-1, provide an interesting array of possible testing locations, particularly for the water erosion, phosphorous and nitrogen contamination indicators, but for the other indicators as well. These sites are also attractive given that the watershed has been used as a base unit of analysis for many of the valuation exercises cited in the literature.

As part of the iterative process of the valuation exercise for the watershed basin and methodological framework for aggregating to the provincial level, IISD would continue to explore the linkages with AAFC's ideas for integrated economic and environmental modelling using CRAM and LUAM.

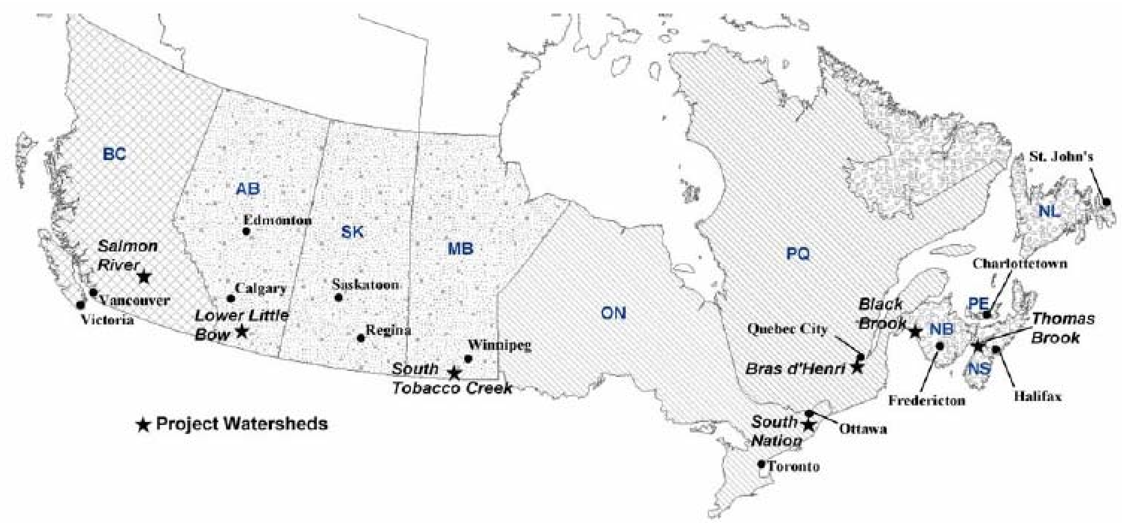

Figure 9-2. Project watersheds for the Watershed Evaluation of Best Management Practices (WEBS) project (from Ducks Unlimited Canada 2004).

Table 9-1. Listing of WEBs best management practices by project site.

Salmon River – British Columbia

1 Off-stream watering
2 Fencing riparian areas
3 Irrigation management
4 Permanent cover

Lower Little Bow Watershed - Alberta

1 Buffer Strip
2 Manure Management
3 Off-Stream Watering with Fencing
4 Conversion from Annual Cropland to Greencover
5 Off-Stream Watering with No Fencing

Steppler - South Tobacco Creek - Manitoba

1. Zero Tillage Practice Compared to a Conventional Tillage Practice
2. Holding Pond to Capture Runoff from a Cattle Containment Area
3. Conversion of Cropped Land to Forage
4. Development or Enhancement of Riparian Area along Water Courses
5. Use of Small Reservoirs to Reduce Downstream Runoff

South Nation – Ontario

1. Restricted Cattle Access (RCA) vs. Unrestricted Cattle Access (URCA)
2. Tile water flow control
3. Nutrient management planning

Bras d'Henri - Québec

1. Reduced herbicide use
2. Slurry management
3. Buffers to reduce runoff velocity
4. Crop rotation

Black Brook Watershed – New Brunswick

1. Variable grade diversions
2. Grassed waterway
3. Vegetated buffer zone

Thomas Brook - Nova Scotia

1. Storm water diversion from farm buildings;
2. Reduced stream access for cattle.
3. Nutrient Management Plans for watershed agricultural lands
4. Retention pond BMP

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