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AGRICULTURAL RIPARIAN HEALTH: THEORY, CONCEPTS AND POTENTIAL INDICATORS

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1. BACKGROUND

This report was produced under the National Agri-Environmental Health Analysis and Reporting Program (NAHARP). It aims to provide the theoretical basis for the development of a “riparian agroecosystem health indicator” as part of a broader project entitled “Agri-environmental Indicators of Landscape Ecology.”

More specifically, the report aims to:

- Define the concept of agricultural riparian zone;
- Demystify the concept of ecosystem health;
- Identify the main components of agricultural riparian health;
- Shed light on the potential relationships between agricultural riparian health and aquatic habitat health;
- Propose health indicators based on the observations made in this report, but also on the literature review that had previously been conducted as part of the project (Christensen *et al.*, 2010);
- Provide a number of possible brief methodological approaches for developing a health indicator;
- Propose possible solutions for assessing the indicator at the agricultural landscape scale;
- Assess how the riparian health indicator is linked to the other agri-environmental indicators of the «Agri-environmental Indicators of Landscape Ecology» project.

This document is a working tool designed to guide discussions regarding indicators that could be used to quantify agricultural riparian health. It is also intended to provoke thought and discussion on the use of the term «health» applied in the specific case of agricultural riparian zones. As we will see, the concept of «health» is relatively vague and subjective. This puts into perspective the idea that the concept of “agricultural riparian health” should emerge from currently available scientific knowledge in the areas of riparian ecology and ecosystem health assessment and also from the knowledge and perceptions of the working group members.

2. WHAT IS A RIPARIAN AGROECOSYSTEM, OR RATHER, AGRICULTURAL RIPARIAN ZONE?

2.1. Definition of ecosystem

The Canadian Encyclopedia defines ecosystem as follows:

> A limited space within which living beings interact with nonliving matter at a high level of interdependence to form an environmental unit is called an ecosystem. On a large scale, ecosystems have been defined on the basis of geographical extent alone (e.g., arctic, tall-grass prairie or hardwood forest). These very large areas are often called biomes. On a smaller scale (e.g., dune or bog) the term refers to spatial dimensions and to precise SOIL conditions within which interactions among components take place (Historica-Dominion Institute, 2010).

Although an ecosystem can have specific boundaries (e.g., banks of a marsh), the boundaries of an ecosystem are sometimes subjectively chosen for practical reasons having to do with the goals of a particular study (University of Michigan, 2010). The concept of ecosystem can appear vague since ecosystems are never isolated, but rather included as part of larger systems (biomes, ecozones) and closely related to adjacent ecosystems (Rapport, 1992). The same is true for the continuum that exists from stream to river, from river to estuary and from estuary to ocean. This continuum also exists between terrestrial and aquatic environments.

2.2. Riparian zones: a continuum to the terrestrial-aquatic boundary

The riparian zone can be defined as the semi-terrestrial space between the upland environment and the aquatic environment within landscapes and watersheds (Vidon et al., 2010) (Figure 1). The riparian zone is an ecotone (Naiman and Décamps, 1997). The concept of ecotone is distinguished from that of ecosystem by the fact that an ecotone is the transition zone between two adjacent ecosystems (Gosz, 1993). An ecotone has characteristics defined by space and time scales and by the strength of interactions between the adjacent ecosystems (Gosz, 1993).

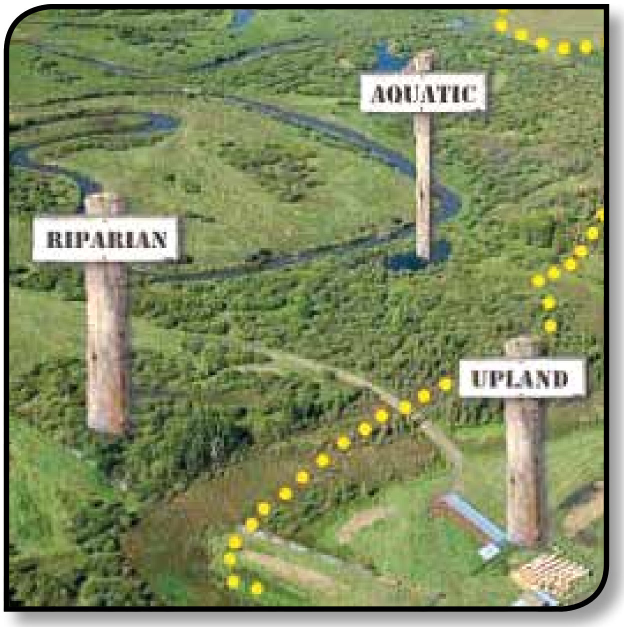

Figure 1: **Delineation of the riparian zone (dotted line) at the boundary between the upland and aquatic environments on the basis of floodplain and gradual transition from vegetation heavily influenced by the presence of water to upland vegetation (Fitch and Ambrose, 2003). Source: Cows and Fish, Alberta Riparian Habitat Management Society. Online at www.cowsandfish.org.**

Riparian zones or riparian ecotones are multidimensional systems shaped by the following basic principles (Naiman *et al.*, 2005):

1. Water saturation gradients are determined by topography, geologic materials, and hydrodynamics.
2. Biophysical processes are driven by dynamic water saturation and energy gradients.
3. Surface and subsurface entities provide feedbacks that control organic energy and material fluxes.
4. Biotic communities are structured or arrayed in space and time along gradients in three dimensions: longitudinal, lateral and vertical.

Riparian zones are characterized by often large, complex biophysical gradients and are structured by geomorphic conditions, flood dynamics and animal activities (Naiman *et al.*, 2005). However, in a human-dominated world, riparian zones must also be viewed as natural–cultural systems in which socioeconomic factors and cultural representations of landscapes shape the management and conservation of riparian areas (Naiman *et al.*, 2005).

Finally, it is important to note that, despite the small area occupied by riparian ecotones in the landscape, they have a major influence on the movement and retention of water, soil particles, nutrients and agricultural inputs (Gregory *et al.*, 1991). Riparian ecotones are now recognized as key elements for biodiversity and biogeochemical "hot spots" within landscapes and watersheds (McClain *et al.*, 2003; Décamps *et al.*, 2004; Naiman *et al.*, 2005).

2.3. Agricultural riparian zones: difficult to define

Agricultural riparian zones are a good example of the effect of socioeconomic and cultural factors on the dynamics of a system. In agricultural areas, the physiognomy of many streams and riparian ecotones has been completely altered by human activities designed to enhance the area and productivity of agricultural land. In Quebec, for example, the creation of drainage channels and the straightening of streams in agricultural areas have altered over 30,000 km of natural streams and have created some 10,000 km of regulated ditches (Beaulieu, 2001). It is estimated that the hydrographic network has doubled in density as a result of these measures (Beaulieu, 2001). Added to that is the loss of over 75% of the wetlands of the St. Lawrence River Valley due to filling or draining (Ducks Unlimited Canada, 2007).

These measures are primarily the result of the replacement of traditional pasture-based agriculture with intensive agriculture focused on hog production and annual crops (corn and soybean) (Pan et al., 1999; Jobin *et al.*, 2003). In the southern St. Lawrence Lowlands, moderate intensification of agriculture occurred between 1981 and 2006, due to the decrease in the proportion of forest, non-productive land, idle land or grasslands or pasture (Huffman and Eilers, 2010). Some agricultural watersheds in this ecozone have undergone extensive forest clearing as a result of agricultural intensification (Bélanger and Grenier, 2002), which often leads to an absence of riparian forests in agricultural landscapes.

In short, the riparian zone of many streams has almost completely disappeared, leaving streams exposed to vertical erosion. Hydrologically and biophysically complex riparian systems (Figure 2) are often replaced by much less complex systems that occupy less space in the watershed (Figure 3). Physical alterations by past human activity make it more difficult to identify riparian zones in agricultural landscapes.

Figure 2: Complex riparian zones dominated by tree species of the sugar maple–yellow birth domain in the Laurentians (source: J. Fortier).

Figure 3: Left, riparian zone of a pasture in the Estrie region (Quebec) that has lost its complexity (source: J. Fortier). Right, aerial photograph of an agricultural riparian zone in Alberta (source: Fitch and Ambrose 2003, online at www.cowsandfish.org). In both cases, it is more difficult to clearly identify where the riparian zone is located compared to Figure 1.

It is also important to determine the type of stream on which agricultural riparian health will be assessed. Should intermittent streams and drainage ditches be included, or should the assessment be limited to permanent streams? This is a fundamental question in agricultural landscapes, because the nature of the stream network has been significantly altered, particularly in terms of its density. The stream network can also vary in density depending on the time of year or on precipitation. For example, a study conducted in five agricultural watersheds in the coastal region of Oregon, which is characterized by rainy winters and hot, dry summers, showed that winter stream network densities were almost 100 times greater than summer stream densities (Wigington *et al.*, 2005) (Figure 4).

This example clearly shows that riparian space in agricultural areas is probably much more extensive than is apparent from the permanent stream network. It is therefore important to determine what constitutes a riparian zone and how to define it within the agricultural ecosystem.

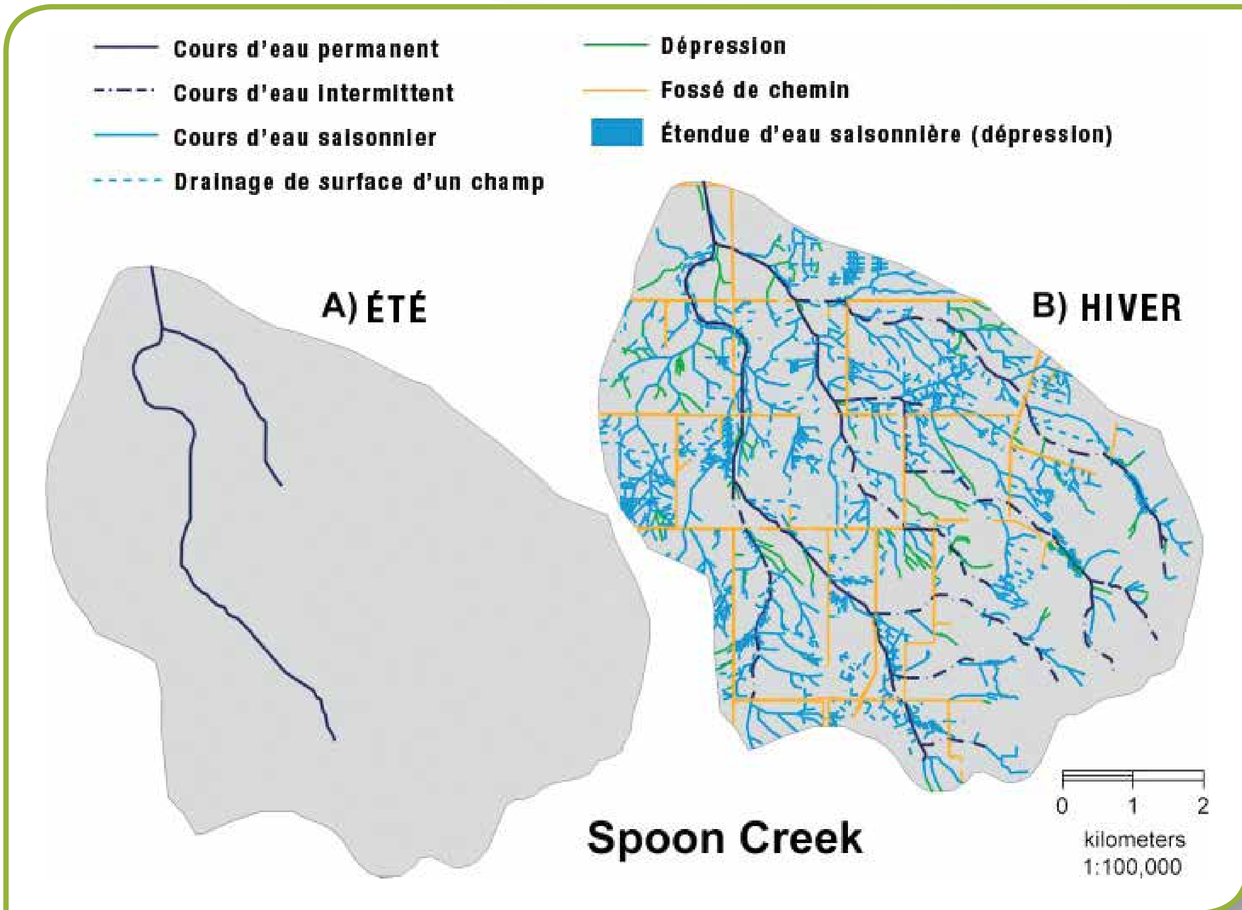

Figure 4: Stream network in the Spoon Creek agricultural watershed in Oregon during the summer of 1997 (A) and winter of 1998-99 (B) (Wigington *et al.*, 2005)

3. ECOSYSTEM HEALTH

The concept of health is generally used to characterize an individual or group of individuals (human population, herd, pack, etc.) (Rapport *et al.*, 1998). In ecology, the concept of ecosystem health has emerged in response to extensive evidence that human-dominated ecosystems are disturbed and dysfunctional (Vitousek *et al.*, 1997). The many disturbances that can affect ecosystems typically lead to their degradation and, consequently, to their inability to generate the same level of ecological services (Cairns, 1997). The ability of the environment to support economic activity and human health is compromised when an ecosystem «becomes sick» (Epstein, 1995; Costanza *et al.*, 1997; McMichael *et al.*, 2006).

3.1. Ecosystem health and stress ecology

The various definitions of ecosystem health are often associated with the concepts of stress ecology (Rapport *et al.*, 1998). When ecosystems are not affected by strong natural or human disturbances, we observe certain well-defined developmental trends. For example, the biomass and sizes of organisms tend to increase and net community production tends to decrease (Odum, 1985). The significant changes observed in ecosystems affected by stress are listed in Appendix 1.

Reponses to stress at the ecosystem level are different from responses at the population level in that they are often more diffuse and longer-term (Odum, 1985). It is therefore important to distinguish between chronic stress, which continues for a long time, and acute stress, which is generally followed by recovery and return to an unstressed state (Odum, 1985).

The identification of indicators (or symptoms) of the response of ecosystems to stress has led to the definition of «ecosystem distress syndrome.» In aquatic ecosystems, the ecosystem stress syndrome comprises the following symptoms (Rapport, 1991):

- alteration in biotic community structure to favour smaller forms
- reduced species diversity
- increased dominance by ‘r’ selected species (opportunistic species)
- increased dominance by exotic species
- shortened food-chain length
- increased disease prevalence
- reduced population stability.

In most cases, disturbed ecosystems exhibit all of the above seven symptoms (Rapport, 1992). The ecosystem distress syndrome provides a useful starting point for assessing ecosystem health, although the above symptoms are too often observed once the process of ecosystem degradation is well advanced (Bormann, 1985).

3.2. Overall index of ecosystem health

On the basis of the concept of ecosystem distress syndrome, Costanza *et al.* (1992), proposed a description of ecosystem health on the basis of the following components:

1. homeostasis
2. low incidence of disease
3. diversity or complexity
4. stability or resilience
5. vigour or scope for growth
6. balance between system components

According to these authors, it is important to consider the six components, or at least the majority of them, simultaneously. Costanza *et al.* (1992) proposed a global ecosystem health index or « health index » (HI) based on three indicators:

$$HI = V^*O^*R$$

***V* = *Ecosystem vigour*.** Vigour is measured in terms of activity, metabolism or primary productivity. An example of reduced vigour is the decline in the abundance of fish and the infertility of agricultural soils within the Great Lakes Basin (Rapport *et al.*, 1998).

O* = *Ecosystem organization. Organization can be assessed as the diversity and number of interactions between system components. For example, the decline in ecosystem organization in the Great Lakes Basin became evident with the abrupt shift in dominance from highly organized nearshore benthic fish associations to the relatively less organized offshore pelagic associations (Rapport *et al.*, 1998).

R* = *Ecosystem resilience. Resilience is measured in terms of a system's capacity to maintain structure and function in the presence of stress. The loss of resilience in the Great Lakes resulted in extinctions of native fish species and a flip to more eutrophic conditions (Rapport *et al.*, 1998). The concept of ecological resilience can also be defined as the amount of disturbance necessary to move an ecosystem from one state to another (Figure 5).

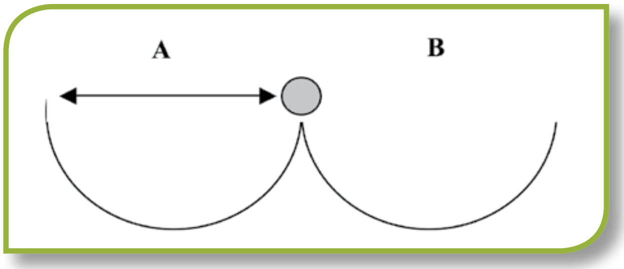

Figure 5: Ecological resilience is the amount of disturbance necessary to change the state of an ecosystem from state A to state B. The "ecological threshold" is the point at which there is an abrupt change in an ecosystem (Groffman *et al.*, 2006).

3.3. Ecosystem health and ecological integrity

Although the concept of ecological integrity is related to that of ecosystem health, there are fundamental differences between the two concepts (Karr, 1996; Jorgensen *et al.*, 2010).

3.3.1. Concept of ecological integrity [1]

The development of self-organizing systems, such as ecosystems, is generally characterized by phases of rapid organization to a steady-state, followed by a period during which the system maintains itself at the new steady state. The change in the state of the system may be continuous (e.g., plant succession in the absence of disturbance) or catastrophic (e.g., desertification).

The change in state is accomplished by the addition of new structures for dissipating intercepted solar energy that consist of (1) new pathways that connect old components or (2) new components and their associated new pathways. Each "spurt" results in the system moving further from equilibrium and occurs when environmental conditions exceed a certain threshold. When environmental conditions become stable, the system enters a new steady state. The path that the system follows as it develops in a stable environment is called the thermodynamic branch (Figure 6).

As ecosystems are driven away from equilibrium, they become more organized and effective at dissipating solar energy. At the same time as this self-organizing process is occurring, external environmental fluctuations are tending to disorganize the system. There is a point in state

[1] This section of the report is taken from Kay (1991).

space where the disorganizing forces of external environmental change and the organization thermodynamic forces are balanced. This point is referred to as the optimum operating point (Figure 6). The climax community in ecological succession is a classic example of an optimum operating point. However, over time, new species will enter the equation, as will new environmental phenomena, and this will result in a new climax community, i.e., a new optimum operating point.

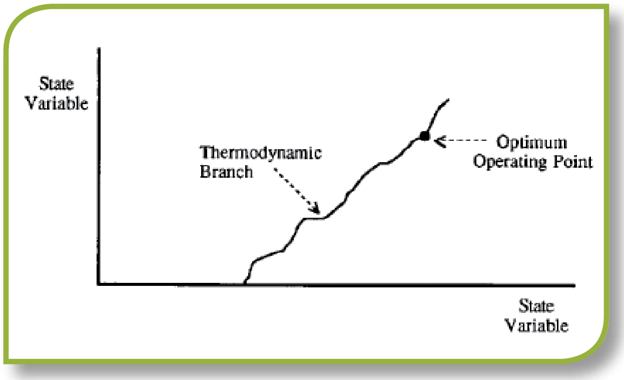

Figure 6: The ecosystem develops along a thermodynamic branch until it reaches an optimum operating point (Kay, 1991). Examples of state variables could be intercepted radiation (x) and biomass (y).

Ecological integrity can therefore be defined as the ability of the system to maintain its organization and to continue its process of self-organization. In other words, it has to do with the ability of the system to attain and maintain its optimum operating point. However, because ecosystems are not static, it is not possible to identify a single organization state of the system that corresponds to integrity. Instead, there is a range of organizational states for which the ecosystem is considered to have integrity. It depends largely on the environment in which the system evolves.

3.3.2. Ecological integrity and the response to environmental change [2]

A sudden change in the environment in which a system develops can have various effects on the organization and integrity of the system:

I. The ecosystem does not move from its original optimum operating point. The integrity of the system is not affected, at least not immediately. For example, a terrestrial ecosystem adapted to flood or drought will not be affected by a short-term flood or drought.

II. The ecosystem moves from its original operating point but returns to it. The system is able to reorganize itself in response to an environmental change and its integrity is preserved. For example, a forest fire is an event that moves a system well away from its optimum operating point, but through natural regeneration, it will eventually return to that point.

III. The system moves permanently from its original optimum operating point.

Case 0: The system collapses. The environmental changes are so drastic that the ecosystem becomes inhospitable and virtually uninhabitable. An example is the process of desertification caused by prolonged drought and wind erosion, which leads to the collapse of the system. Dead lakes caused by acid rain from mining in the Sudbury region are another example. In these examples, some life forms generally continue to persist in the system, but a complex ecosystem does not.

Case 1: The system remains on one of its original thermodynamic branches (Figure 7). In response to environmental changes, the ecosystem maintains its original set of dissipative structures or moves back to some set that represents an earlier stage in development. The level of operation of the individual structures has changed (sometimes even catastrophically), but certain structures of

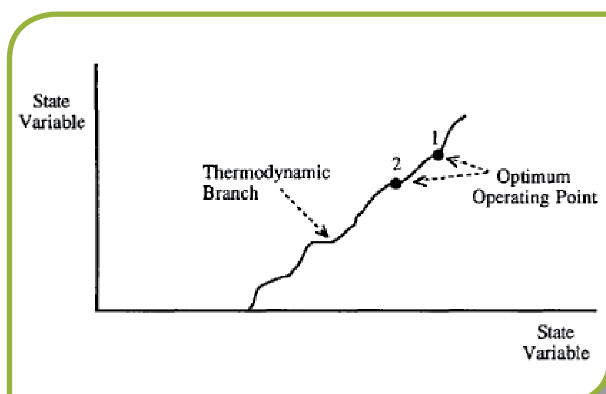

Figure 7: In response to an environmental change, the thermodynamic branch of the ecosystem moves from its optimum operating point (1) to a new optimum operating point (2) (Kay, 1991).

[2] This section of the report is taken from Kay (1991).

the ecosystem remain recognizable. Even if the environmental conditions return to their original state, it is not a given that the system will return to the original optimum operating point. For example, maple forests subjected to acid rain are shifted to a state of less productivity and lower biomass, but it is not known whether these ecosystems will recover their functions in the absence of acid rain. In this case, the system's integrity is affected, because its organization and function are affected.

Case 2: System bifurcates to a new thermodynamic path (Figure 8). In response to environmental change, new dissipative structures are added to the system and/or some of the original ones disappear. The new structures can be new pathways for energy flow and the level of operation of the system is slightly changed. As a result of this permanent addition of new structures, the system bifurcates to a new optimum operating point. The ecosystem is unlikely to return to its former state without human intervention. The new ecosystem's organization is not completely different from the original, but its integrity is affected in the sense that the organization has been permanently altered. The introduction of exotic species into a lake is a good example of this type of ecosystem response. New species associations (dissipative structures) occur, despite the fact that the system still has many similarities to the original system.

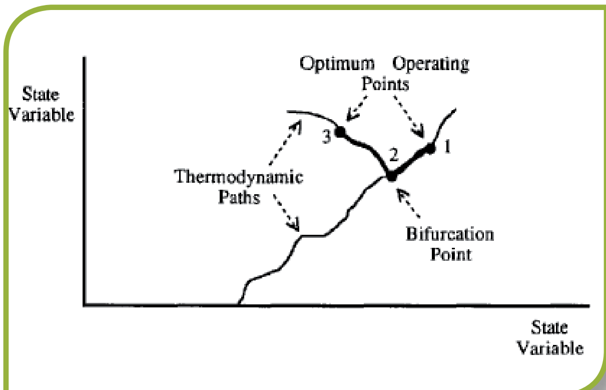

Figure 8: In response to changing environmental conditions, the thermodynamic path of the system moves away from its optimum operating point (1) through a bifurcation point (2) and then to a new optimum operating point (3) (Kay, 1991).

Case 3: The system moves to a new thermodynamic branch (Figure 9). In response to environmental changes, the system undergoes a catastrophic change that leaves the system so reorganized that there is no possibility of the system returning to its original optimum operating point, even if the environmental conditions return to their original state. The integrity of the system is affected, as the structure and composition of the system have been modified. For example, in some tropical ecosystems, clear-cutting has resulted in soil erosion so severe as to effectively change the soil type. It is therefore unlikely that the original system will reappear since the soil properties have been completely altered.

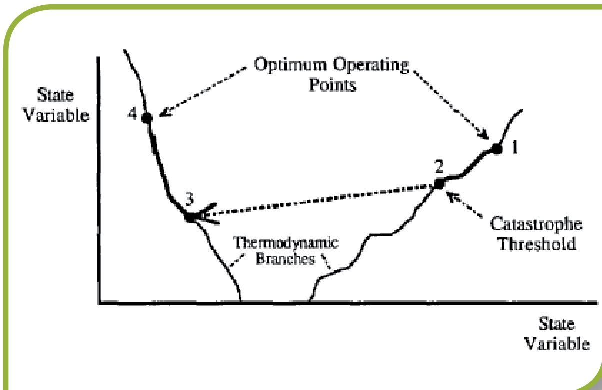

Figure 9: The environmental change drives the ecosystem from its optimum operating point (1) through a catastrophe threshold (2) to a new thermodynamic branch (3) and eventually to a new optimum operating point (4) (Kay, 1991).

How do we view the ecological integrity of an ecosystem that is reorganizing itself in the face of environmental change? It could be argued that an ecosystem whose optimum operating point has been permanently altered loses ecological integrity (cases 0 to 3). It could also be argued that any system that can maintain itself at an optimum operating point, regardless of whether it is its original point or not, conserves its ecological integrity. This means that an ecosystem that bifurcates from its thermodynamic path could be considered to have integrity if it reaches a new optimum operating point.

Between these two extremes, there is the possibility of defining some optimum operating points as being undesirable changes in the system and therefore representing a loss of integrity. At the same time, other optimum operating points could be considered desirable. An ecosystem

[2] Cette section du rapport a été tirée de Kay (1991)

that maintains its new optimum operating point or moves towards it could be qualified as «healthy.» Table 1 summarizes the possible responses of an ecosystem to environmental change and the issues concerning integrity that should be taken into consideration, particularly by land use managers.

Table 1: Possible responses of an ecosystem to environmental change

I. The ecosystem does not move from its original optimum operating point.

II. The ecosystem moves from its original optimum operating point but returns to it.

Issues concerning integrity to be considered:
(1) How far is the system moved from its optimum operating point?
(2) How long will it take to return to its optimum operating point?
(3) What is the stability of the system upon its return?

III. The system moves permanently from its original operating point.

Case 0: The system collapses.
Case 1: The system remains on its original thermodynamic branch (Figure 7).
Case 2: The system bifurcates to a new thermodynamic path (Figure 8).
Case 3: The system moves to a new thermodynamic branch (Figure 9).

Issues concerning integrity to be considered:
(1) How far is the new optimum operating point from the old?
(2) How long does it take to reach the new optimum operating point?
(3) What is the stability of the system at the new optimum operating point?
(4) If the environmental conditions return to their original state, will the system return to the original optimum operating point?

3.3.3. What is the difference between ecosystem health and ecological integrity?

Ecological integrity can refer to the ability of the ecosystem to reach its optimum operating point and to maintain it in the face of an environmental change (Kay, 1991). However, ecological integrity often refers to the original conditions that existed in an ecosystem before it was altered by humans (Karr, 1996).

Biological integrity refers to the capacity to support and maintain a balanced, integrated, adaptive biological system having the full range of elements (genes, species, assemblages) and processes (mutation, demography, biotic interactions, nutrient and energy dynamics and metapopulation processes) expected in the natural habitat of a region (Karr, 1996). For example, in many parks and nature reserves, a proportion of the area is set aside for the protection and restoration of ecological integrity.

Following a natural or human disturbance, an ecosystem can undergo major changes that make it impossible to restore ecological integrity (Figure 8 and Figure 9). This is true of a number of riparian systems in agricultural landscapes, which undergo numerous stresses on an annual basis (tillage, pesticide application, trampling by cattle, etc.), in addition to past isolated stresses (drainage, shoreline straightening, regulation, flow regulation) (section 4.4). The combined effect of these stresses makes it virtually impossible to go back. When ecological integrity cannot be preserved or restored, ecosystem health is ideally the objective (Karr, 1996).

The concept of ecosystem health or ecological health applies to already degraded, disturbed or human-dominated ecosystems (Karr, 1996). An ecosystem that was previously or is currently stressed can still be resilient and maintain its organization. A healthy ecosystem is capable of achieving and maintaining a new optimum operating point, and of returning to that point following stress (Figure 8 and 9). In a healthy agroecosystem, it is therefore assumed that there is little nonpoint source pollution, a low level of greenhouse gas production, little soil erosion, pollutant retention and transformation, habitats for plants and animal life, etc. In other words, the health of an ecosystem can refer to its capacity to generate diverse, high-quality ecological services over time.

In the case of human-disturbed ecosystems, it is important to take certain dimensions of ecological integrity into account. It is legitimate to ask to what point the stressed system is different from the original system or from a system that most closely resembles it today. The use of certain reference conditions (integrated or minimally disturbed natural ecosystems) can be desirable to identify the desirable or undesirable states and components of the ecosystem (Norris and Thoms, 1999). It is important to bear in mind that the quality and diversity of the ecological services provided by an agricultural riparian zone will depend on the optimum operating point that is eventually reached.

3.4. Sociocultural dimension of ecosystem health

In contexts where there may be more than one optimum operating point for a given ecosystem, the identification of a desired state of health is largely subjective (Kay, 1991; Rapport, 1992). As a result, the concept of ecosystem health has an important social dimension in that it can depend on the knowledge, priorities, objectives, values and perceptions of individuals and organizations (Norris and Thoms, 1999). The players involved in the management of a natural system must agree on the desirable or undesirable states and must determine the main ecosystem services that must be supported by a healthy riparian zone in a given socioeconomic and cultural context (Karr, 1996).

Ecosystem services (including ecosystem goods) are defined as the benefits people obtain from ecosystems (Millennium Ecosystem Assessment, 2005). They include not only benefits recognized by the public, but also benefits of which the public is unaware (Costanza *et al.*, 1997). Ecosystem services have an ecological, sociocultural and economic value (or dimension). The recognition of the value of ecosystem services therefore depends largely on the perception of individuals and organizations involved, which influences the choice of the state of health desired for a natural system (Rapport *et al.*, 1998). As part of this project, which is aimed at developing a Canada-wide indicator, it is important to bear in mind that the concept of agricultural riparian health could vary from region to region or from province to province depending on the perceptions, objectives and culture of the players involved in the management of the riparian zone (Nassauer, 1995).

3.5. Critique of the concept of ecosystem health

Although some researchers and land use managers have adopted the concept of ecosystem health, it has not been unanimously accepted within the scientific community. For example, Suter (1993) sees many operational problems with the use of the concept as a management criterion. According to Suter, health is not an operational, measurable property of a system, but rather an abstract image or metaphor. The health index developed by Costanza *et al.* (1992), which integrates system vigour, organization and resilience, appears to have a number of weaknesses. According to Suter (1993):

- Indices of heterogeneous variables are generally ambiguous. It is difficult to know why such indexes are high or low. If an index is low, it may be because all components are slightly damaged or because one component is severely damaged. In indices of heterogeneous variables, low values of one component can be eclipsed by high values of another.
- The distribution functions and other statistical properties of indices of heterogeneous variables are difficult to define and are often determined arbitrarily.
- Indices of heterogeneous variables are not a measure of any real-world property. The units that characterize such indices are nonsense from a scientific point of view.
- Combining heterogeneous measures into a single index implies that there is only a single, linear scale of response by ecosystems to disturbance and only one mode of action in response to disturbance.
- Indices of ecosystem health cannot be tested in a controlled study in the laboratory. This means that correlations between an health index value and certain variables measured in the field can never be verified in controlled studies.

- The health index HI = V*O*R produces nonsense results. For example, when a lake moves from an oligotrophic to a mesotrophic state, vigour increases (increase in primary productivity) and organization increases (increase in diversity), which means that the HI also increases. However, can we really say that a nutrient-enriched lake with its loss of clarity is healthier, whereas it is showing signs of eutrophication?

In short, in the development of an indicator of agricultural riparian health, it will be critical to identify the components of riparian health in such a way that they do not lead to a vague interpretation of the state of health. Given that the agricultural riparian health indicator will be used as a tool for managing the agricultural riparian zone, it must allow for a relatively precise diagnosis. It must also allow for the identification of the causes of the measured state of health, and the definition of appropriate riparian zone management and restoration measures.

Sites considered degraded or high priority could be identified using the light detection and ranging (LiDAR) approach that will be developed. Field teams may be sent on site to observe and assess the extent of the restoration work to be carried out at degraded sites.

4. COMPONENTS OF AGRICULTURAL RIPARIAN HEALTH

Given that the agricultural riparian zone is an environment that is often highly disturbed by human activities, it would be unrealistic to consider restoring the ecological integrity of all riparian systems to what it once was in the natural state. However, certain less disturbed agro-riparian ecotones may have the capacity to return to their original optimum operating point, or at least close to it, if protection and restoration efforts are undertaken. The objective here is not to provide a detailed description of all structures and functions of the riparian zone that are required to ensure that the ecological integrity of the system is maintained. However, natural, integral riparian systems can be used to identify components with which important ecosystem functions and services are associated. A number of indicators of riparian health are identified in this section. They will provide a theoretical basis for the proposal of potential health indicators described in section 5 of the report.

4.1. Biophysical connectivity

The ecological functions that make the riparian zone a distinct system in the agricultural landscape are largely dependent on a high level of connectivity and interaction between riparian vegetation, soil, groundwater and surface water (Lowrance *et al.*, 1997; Naiman *et al.*, 2005) (Figure 10). As will be explained later, the state of this connectivity between the biological and physical elements of the landscape depend on natural factors, such as geomorphology, but also on human disturbances, particularly hydrologic alteration (channelization, drainage, deforestation).

4.1.1. Lateral and vertical connectivity

Many functions, both hydrologic and biogeochemical, are attributed to woody riparian vegetation. These functions are largely dependent on vertical and lateral interactions between vegetation, soil water,

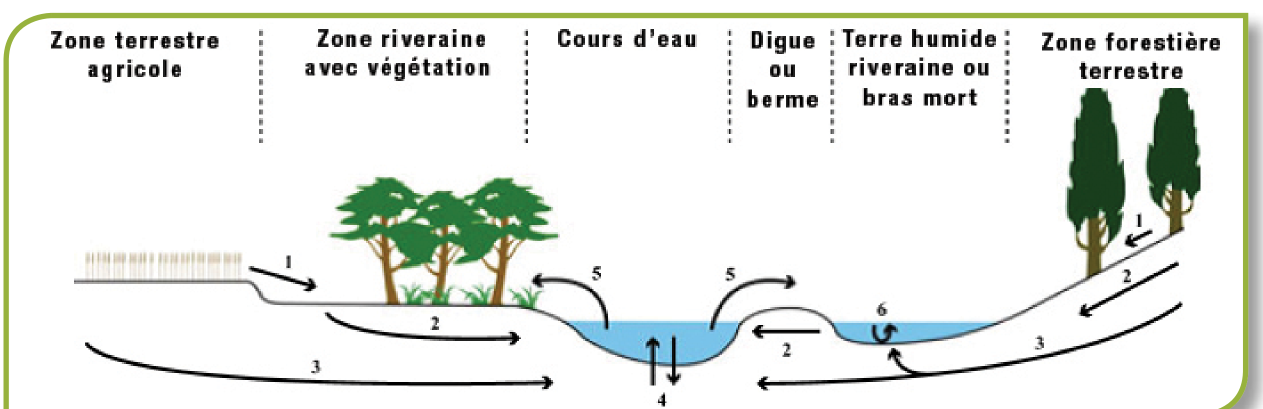

Figure 10: **Complex hydrologic dynamics and biophysical interactions at the interface of upland and aquatic environments in agricultural and forested landscapes (Vidon et al., 2010). Dominant flows are indicated: (1) overland flow (2) subsurface flow; (3) deep groundwater flow; (4) hyporheic flow; (5) overland flow; (6) mixing.**

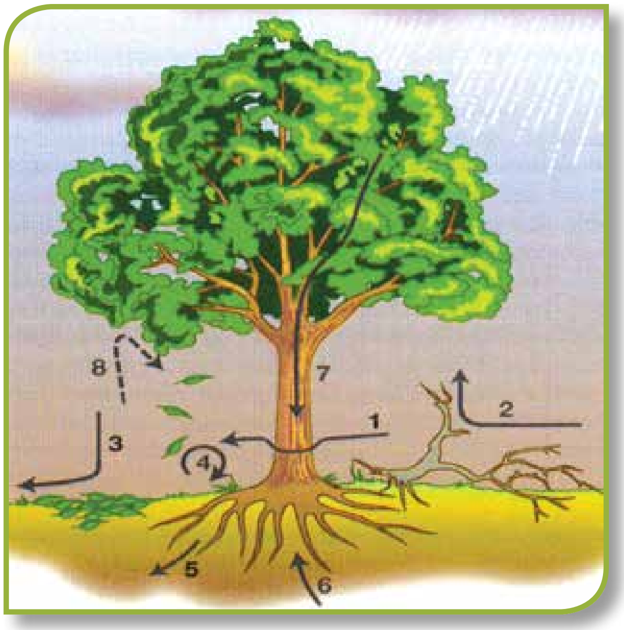

Figure 11: **Main physical impacts of woody riparian vegetation on hydrologic processes: (1) interaction between overbank flow and stems, branches and leaves of trees reduces current velocity; (2) dissipation of current energy by woody debris accumulations in streams and riparian zones; (3) change in the infiltration rate of flood waters and rainfall by litter; (4) increase of turbulence as a consequence of root exposure; (5) increase of substrate macroporosity by roots; (6) increase of the capillary fringe by fine roots; (7) stemflow (the concentration of rainfall by leaves, braches and stems); (8) condensation of atmospheric water and interception of dew by leaves and branches (Tabacchi *et al.*, 2000; Naiman et al., 2005).**

soil microorganisms and surface water. Hydrologic interactions between riparian vegetation and streams depend on the state of lateral hydrologic and biophysical connectivity between the stream, its floodplain and the riparian vegetation. For example, riparian trees have a significant influence on controlling overbank flows (Figure 11) (Tabacchi *et al.*, 2000), but only when streams overflowing their banks can come into contact with floodplain vegetation. On incised streams that do not have a floodplain, the hydraulic energy will remain confined within the channel, streamflow will increase and the banks will be subject to more powerful erosive forces (Sweeney *et al.*, 2004). The existing riparian system will have difficulty effectively providing the bank stabilization, erosion reduction and sediment retention services normally attributed to riparian vegetation (Beeson and Doyle, 1995; Simon and Collison, 2002). In addition, woody debris that has fallen to the ground will have difficulty reaching streams that remain confined within a restricted channel. Woody debris plays an important role in flood control, sediment retention, denitrification and nutrient retention within the stream (Cooke and White, 1987; Tabacchi *et al.*, 2000; Kane et al., 2009). Woody debris, particularly large woody debris, is also an important component of aquatic habitat (Abbe and Montgomery, 1996).

In short, a number of ecosystem services provided by streams and riparian zones are lost when there is clearing of the riparian ecotone and vertical erosion of the stream (Lowrance *et al*., 1997; Sweeney *et al.*, 2004).

Lateral hydrologic connectivity also has a significant effect on certain biogeochemical processes that occur in riparian soils. Kaushal *et al.* (2008) clearly illustrate the importance of this hydrologic connectivity for denitrification in riparian soils. The authors compared the denitrification rates on a given stream in four types of riparian sites: (1) sites where connectivity between the stream and floodplain was restored as well as the riparian vegetation and banks; (2) site where connectivity was not restored, but where riparian vegetation was restored; (3) site where vegetation was not restored, but where there is hydrologic connectivity; (4) unrestored site without connectivity where the stream remains incised (Figure 12). The authors conclude that the riparian zones that have low banks that are connected to the stream, regardless of whether or not they are restored in terms of vegetation composition, are more susceptible to nitrogen denitrification. However, a restoration strategy combining the two approaches (improvement of connectivity and stream restoration) is advisable. Roley *et al.* (2012) reached similar conclusions in an agricultural watershed in Indiana: denitrification increased when the banks were flooded, particularly in the presence of vegetation, which shows the importance of the stream having access to the floodplain.

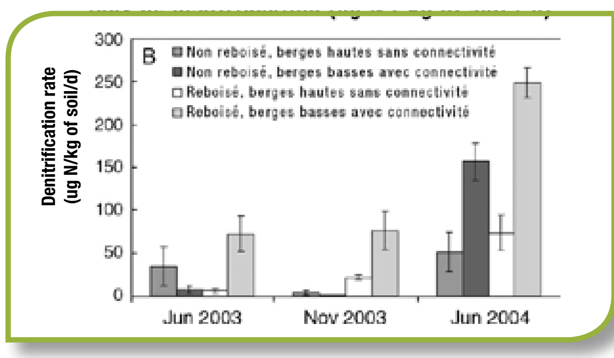

Figure 12: **Denitrification rates observed at four different riparian sites, along a stream in Baltimore, Maryland (Kaushal et al., 2008).**

It is important to note that the degree of lateral hydrologic and biophysical connectivity can vary significantly according to stream type and longitudinal slope gradient (Figure 13). Hydrologic connectivity is influenced by both natural factors and human activities, such as stream straightening and channelization. Several authors suggest that riparian buffer strips along first-order streams (small tributaries) have the highest potential for filtering groundwater and trapping sediments because water saturation conditions favourable to the reduction of nonpoint source pollution are generally observed when there is hydrologic connectivity between the riparian zone and small streams (Dosskey, 2001; Burkart *et al.*, 2004; Sweeney and Blaine, 2007; Tomer *et al.*, 2009).

In North America, small headwater streams can comprise up to 85% of the total length of the stream network and can have a very important effect in terms of nitrogen retention and denitrification (Peterson *et al.*, 2001).

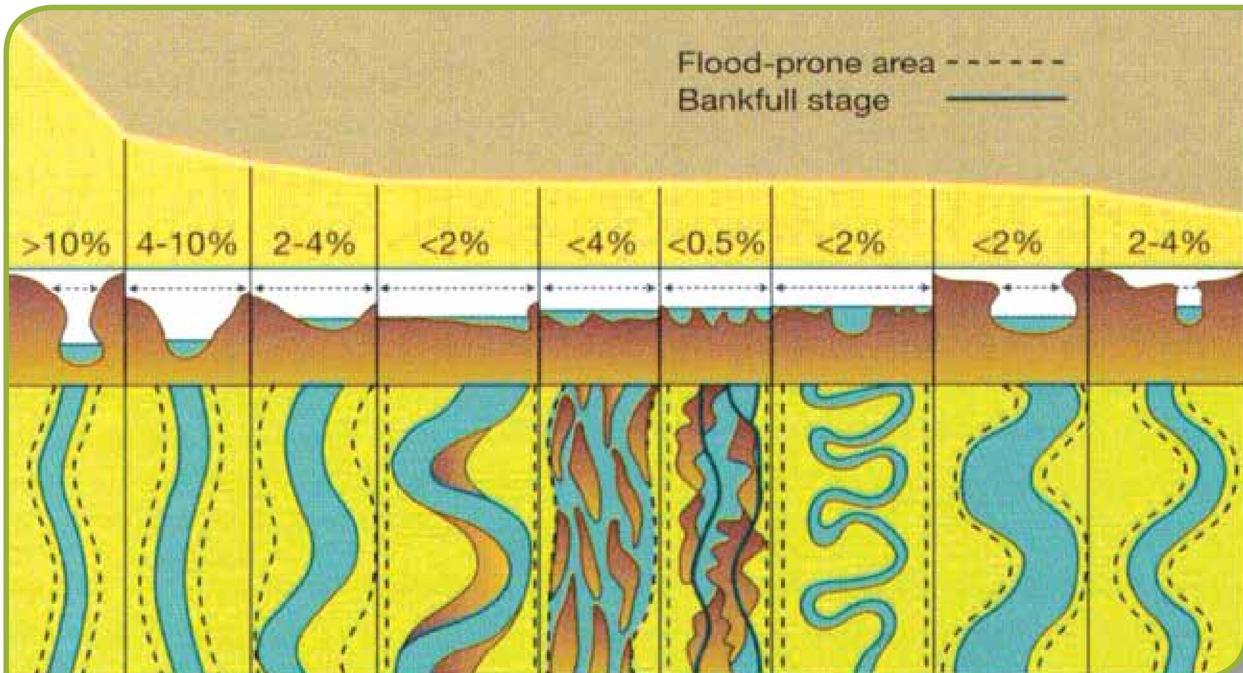

Figure 13: From top to bottom, longitudinal, cross-sectional and plan views of major stream types based on their geomorphological characteristics (Rosgen, 1994; Naiman *et al.*, 2005).

Figure 14 : Example of forested riparian buffers (background) overridden by the intermittent stream network following heavy rains (J. Fortier).

Figure 15: **Intermittent streams can be a significant source of agricultural nonpoint source pollution (J. Fortier).**

Connectivity between agricultural cropland, the intermittent stream network and the permanent stream network is another important aspect (Wigington *et al.*, 2005; Baker *et al.*, 2007). As is often the case in agricultural areas, riparian buffer strips are generally established on permanent streams, which means they can be easily bypassed by the intermittent stream network during episodes of heavy rain or snowmelt (Figure 14). Connectivity between the intermittent and permanent stream network is particularly important in agricultural landscapes, since a major part of nonpoint source pollution comes from small intermittent streams, regardless of whether they are natural streams or were created for drainage purposes (Figure 15).

In agricultural areas, the topography can also be such that there are preferential surface and subsurface flow paths (Tomer *et al.*, 2009) (Figure 16). Soil type also has an important influence on water table movement (e.g., soil hydraulic conductivity) and infiltration of runoff water (soil porosity) (Dosskey *et al.*, 2006). The area of agricultural land drained towards the riparian zone as well as the intensity of the agricultural practices (annual crop vs pasture) are additional factors that affect the extent of hydrologic connectivity between the agricultural area and the riparian zone (Dosskey *et al.*, 2008).

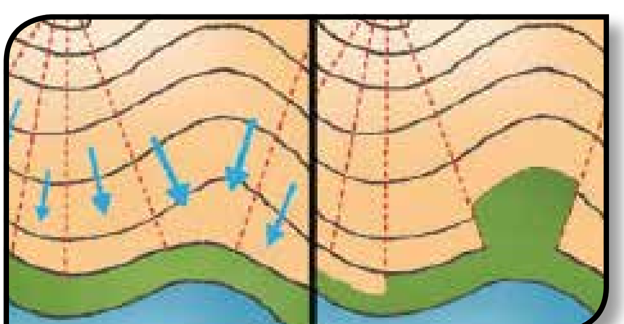

Figure 16: **At left, map illustrating the significance of runoff loads as a function of topography (left). The size of the arrows indicates the significance of the runoff load. At right, development of the riparian buffer (in green) as a function of runoff patterns (Bentrup, 2008).**

The capacity of riparian vegetation to remove nutrients in the water table also depends on vertical connectivity between the riparian vegetation and the water table. The interaction of the vegetation root zone with the water table varies depending on the local stream geomorphology (Figure 17). The less interaction there is between the root zone and the water table, the less the potential for nonpoint source pollution control (Lowrance *et al.*, 1997). This interaction is generally the weakest on incised streams.

Vertical connectivity between vegetation and the water table can also depend on the type of vegetation present. Vegetation with a deep root system can increase denitrification and nutrient removal in deep soil layers (Gift *et al.*, 2008; Dosskey *et al*., 2010).

The width of developed or uncultivated riparian zones also has an effect on lateral and vertical hydrological connectivity. The protection of both water quality and aquatic and terrestrial habitat quality increases with the width of the riparian buffer when there is vertical and lateral interaction

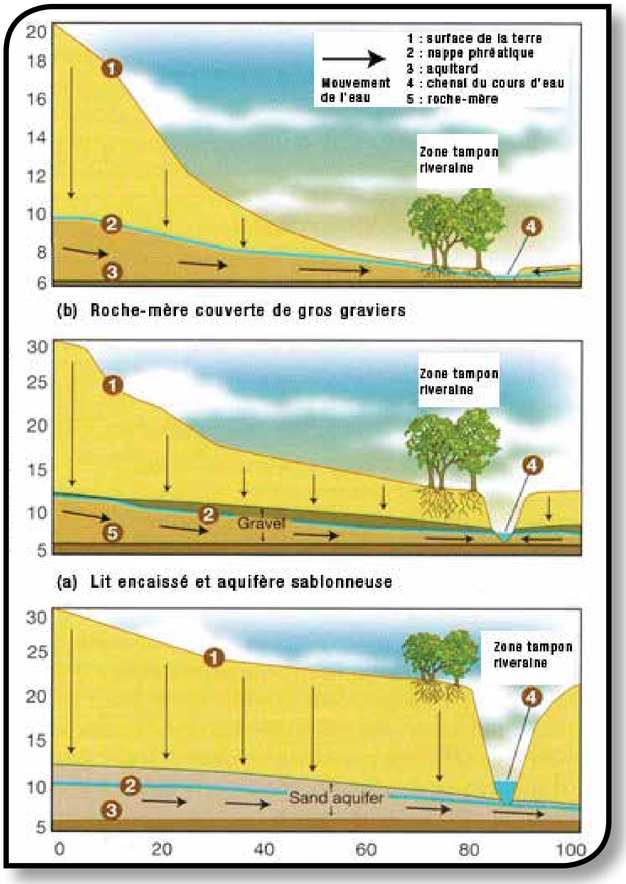

Figure 17: Depending on the local stream geomorphology, riparian vegetation will have varying degrees of interaction with subsurface water moving toward the channel. In (a) and (b) (bedrock overlaid with coarse gravel), the riparian vegetation has opportunities to interact with subsurface water, unlike in (c) (deeply incised channel and sand aquifer (adapted from Lowrance *et al.*, 1997 by Naiman *et al.*, 2005).

between the riparian vegetation, stream and water table (Bentrup, 2008; Dosskey *et al.*, 2008). Other factors can be used to determine the optimal buffer width on the basis of the ecosystem services to be maintained or generated (e.g., soil type, type of agriculture, stream type, topography, threatened species) (Bentrup, 2008; Dosskey *et al.*, 2008).

The loss of connectivity between incised streams and their floodplains can be detrimental to both terrestrial and aquatic biodiversity, particularly for species that grow, nest and feed in the riparian zone and stream (wetland plants, aquatic birds, amphibians, benthic invertebrates, fish, etc.) (Quinn *et al.*, 1992; Barton, 1996; Toth *et al.*, 1998; Sweeney *et al.*, 2004; Baxter *et al.*, 2005). Biophysical connectivity between riparian vegetation and streams is particularly important to terrestrial and aquatic wildlife communities because reciprocal flows of invertebrate prey link terrestrial and aquatic food webs (Baxter et al., 2005).

4.1.2. Terrestrial connectivity

Longitudinal terrestrial connectivity is a spatial relationship between the natural systems of a matrix dominated by agriculture (Figure 18). Riparian zones can serve as corridors to facilitate the movement and dispersal of terrestrial plant and animal species within agricultural landscapes (Spackman and Hughes, 1995; Burbrink *et al.*, 1998).

Figure 18: Example of a riparian corridor connected to a forest patch and a windbreak system in the agricultural matrix (USDA).

Corridor size (width and length), continuity and plant composition influence the species that use the corridor (Kringen and Gray, 1990; Spackman and Hughes, 1995; Maisonneuve and Rioux, 2001; Deschênes et al., 2003; Jobin *et al.*, 2004). For example, Spackmann and Hughes (1995) observed that minimum corridor widths of 10 to 30 m are needed to protect 90% of the wetland plants along small streams, and that minimum corridor widths of 75 to 175 m are needed to protect bird species. Environment Canada, in its *Framework for Guiding Habitat Rehabilitation in Great Lakes Areas of Concern*, recommends that 75% of stream length should be naturally vegetated and that there should be a minimum 30-m-wide naturally vegetated adjacent lands area on each side of the stream (Environment Canada, 2004). Additional recommendations are presented for the enhancement of wetlands and forests, particularly in terms of the width of forested corridors, which should be a minimum of 50 to 100 m in width to facilitate species movement.

4.2. Vegetation cover

4.2.1. Ecological functions of riparian vegetation

Certain functions are met more or less effectively depending on the type of vegetation cover in the riparian zone. The table in Appendix 2 presents an overview of the capacity of woody and herbaceous vegetation to fulfil certain ecological functions to which ecosystem services are linked. It can be seen from this table that the respective contribution of woody and herbaceous plants to the various ecosystem services differs. Woody vegetation appears to generate more services, in terms of water and habitat (terrestrial and aquatic) quality protection and at the agronomic level.

4.2.2. Vegetation of natural riparian ecotones in Canada

To assess the health of agricultural riparian zones in agricultural landscapes, it is important to identify the main plant species that naturally colonize the riparian zone in the absence of human disturbance. With this knowledge, it is possible to determine how far a riparian zone that is disturbed by agricultural activities has moved from its natural trajectory in terms of plant succession (Kay, 1991).

In Canada, agricultural activities are concentrated primarily in the southern part of the country, in the following forest regions: Acadian, Great Lakes–St. Lawrence, Deciduous, Boreal, Boreal (forest and grass) and Coast (Figure 19). Most Canadian riparian ecotones currently present in agricultural areas were initially dominated by forests. Even in the Prairies, where herbaceous vegetation dominates the landscape due to the dry climate, it was trees that generally colonized riparian zones. Riparian forests are one of the few forested sites in the Prairies because there is sufficient water for the trees to thrive (Floate, 2004). These riparian forests are of particular importance to the biodiversity of this ecozone (Finch and Ruggiero, 1993; Holloway and Barclay, 2000). However, the soil moisture may not be sufficient to naturally support trees on small streams (Figure 20). Table 2 provides a list of the main native woody species found in those natural riparian zones of Canada in which agricultural activities are currently carried out.

Figure 19: Forest Regions of Canada (Canadian Forest Service, Natural Resources Canada).

Figure 20: **Example of agricultural riparian zone dominated by herbaceous plants and shrubs in the Canadian Prairies (Shelterbelt Centre, Indian Head, Saskatchewan) (photo credit: J. Fortier).**

Table 2 : **Main native tree and shrub species of natural riparian zones in Canada, by region (Gagnon and Bradfield, 1987; Cordes *et al.*, 1997; Gauthier, 1997; Dickmann, 2001; Floate, 2004; Farrar, 2006; Lea, 2008; Waters *et al.*, 2008).**

Region of Canada	Latin name	Common name
Eastern Canada	*Acer Rubrum* *Acer saccharinum* *Alnus incana subsp. rugosa* *Betula papyfera* *Cornus stolonifera* *Fraxinus nigra* *Fraxinus pennsylvanica* *Larix laricina* *Myrica gale* *Picea glauca* *Picea mariana* *Pinus strobus* *Populus balsamifera* *Populus deltoides* *Populus x Jackii (BxD)* *Quercus bicolor* *Salix sp.* *Thuja occidentalis* *Ulmus americana*	Red maple Silver maple Speckled alder White birch Red-osier dogwood Black ash Red ash Tamarack Sweet gale White spruce Black spruce White pine Balsam poplar Eastern cottonwood Jackii poplar Swamp white oak Willow Eastern white cedar American elm
Prairies	*Acer negundo* *Cornus stolonifera* *Fraxinus pennsylvanica* *Larix laricina* *Picea glauca* *Populus angustifolia (A)* *Populus balsamifera (B)* *Populus deltoides (D)* *Populus x jackii (BxD)* *Populus x acuminata (AxD)* *Populus (BxA)* *Populus (BxAxD)* *Salix sp.*	Manitoba maple Red-osier dogwood Red ash Tamarack White spruce Narrowleaf cottonwood Balsam poplar Plains cottonwood Jackii poplar Lanceleaf cottonwood Hybrid poplar Hybrid poplar Hybrid poplar

Region of Canada	Latin name	Common name
Okanagan Valley	*Betula occidentalis* *Cornus stolonifera* *Populus trichocarpa* *Salix sp.*	Water birch Red-osier dogwood Black cottonwood Willow
West Coast	*Abies amabilis* *Acer circinatum* *Acer macrophyllum* *Alnus rubra* *Alnus sinuata* *Picea sitchensis* *Populus trichocarpa* *Pseudotstuga menziesii* *Salix sp.* *Thuja plicata* *Tsuga heterophylla*	Amabilis fir Vine maple Bigleaf maple Red alder Sitka alder Sitka spruce Black cottonwood Douglas-fir Willow Western red cedar Western Hemlock

It is important to note that following the colonization of Canada, the original forests disappeared from most of the areas on which intensive agriculture is now carried out (Bélanger and Grenier, 2002). As a result, today's natural riparian forests likely contain more early or mid-successional species than they originally did. For example, on the West Coast where large riparian conifers were cut down, the woody vegetation often consists of Populus trichocarpa, Acer macrophyllum and Alnus rubra, all early successional species (D. Gagnon, pers. comm.).

Natural riparian zones support a number of threatened and vulnerable plant species. In Quebec, close to half of the 375 species that are threatened, vulnerable or likely to be vulnerable are associated with wetlands or riparian zones (Government of Quebec, 2007).

At the landscape scale, the various orders of streams allows for the development of structurally and floristically distinct riparian systems (Gregory *et al.*, 1991). This is due to variations in hydrologic and geomorphologic conditions from one order to another, such as flood frequency, duration and magnitude (Bendix and Hupp, 2000).

The successional development of riparian vegetation is determined in part by disturbances from floods (Tabacchi *et al*., 1998). In the Prairies, the natural dynamics of riparian forests depends largely on floods. Major floods are essential to the colonization of the riparian zone by poplars, which take advantage of the temporary soil moisture to become established (Cordes *et al.*, 1997).

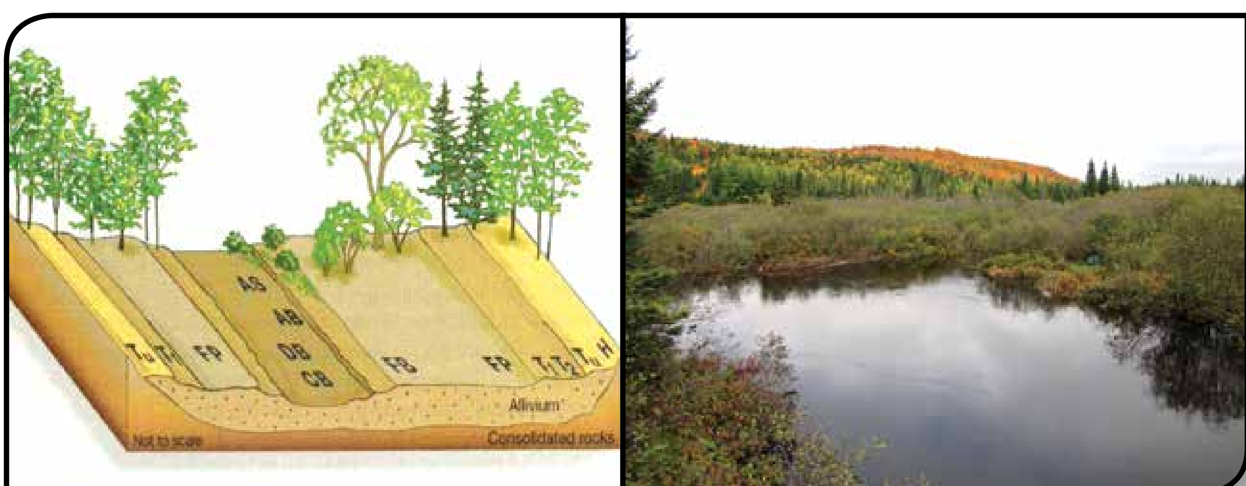

Figure 21: (left) Alluvial landforms: CB-channel bed, DB-depositional bar, AB-channel-shelf bank, FB-floodplain bank, FP-floodplain, T1-lower terrace, T2-upper terrace, H-hillslope (Naiman *et al.*, 2005). Example of a lateral gradient of species, with alder growing in the floodplain, white spruce on terraces and deciduous species on hills (photo credit: J. Fortier).

Figure 22: **Natural riparian zone (sugar maple – yellow birch domain) colonized primarily by herbaceous species (photo credit: J. Fortier).**

The existence of a lateral water saturation gradient (which varies depending on the alluvial deposits) also plays an important role in the distribution of plant species (Figure 21). It is difficult for trees to colonize riparian zones that have a relatively wide floodplain and that remain almost permanently saturated due to soil anoxia (Figure 22).

The invasion of natural (and human-altered) riparian corridors by introduced (exotic) plants is a global phenomenon, in both forests and natural grasslands (Pysek and Prach, 1993; Planty-Tabacchi et al., 1996; Stohlgren *et al.*, 1998; Hood and Naiman, 2000). This phenomenon is due primarily to (1) the proximity of the stream, which is conducive to the transport of propagules to a site where germination is facilitated; (2) the frequency and intensity of disturbances associated with floods, which create an environment where plant competition is reduced; and (3) the continuous availability of water in the soil, which promotes growth and reproduction (Hood and Naiman, 2000). The results of Planty-Tabacchi *et al.* (1996) suggest that the invasion of riparian communities is directly proportional to community richness, regardless of site. These authors suggest that young or more highly disturbed riparian communities have the highest rates of invasion by introduced species. Riparian patches having high edge-to-area (perimeter/area) ratios are also more susceptible to invasion by exotic plants (Planty-Tabacchi *et al.*, 1996). In addition, in agricultural areas, riparian zones are subject to additional disturbances associated with human activities on the periphery of the system. Invasion pressure by exotic plants is higher due to the presence of weeds in adjacent crops and to regulation work, which create environments favourable to invasion (bare soil, erosion, sedimentation).

4.2.3. Vegetation of riparian ecotones in agricultural landscapes

As just mentioned, the type of vegetation that colonizes riparian zones is largely dependent on the state of biophysical connectivity, disturbances related to flooding and geomorphology (Bendix and Hupp, 2000). The dynamics of riparian plant communities is also affected by adjacent agricultural activities and the impacts of past and present activities.

Figure 23: **Large population of common reed along a road ditch (right) and agricultural drainage ditch in Montérégie (left) (photo credit: J. Fortier).**

Deforestation of riparian zones for agriculture is a major disturbance that can cause significant changes. Some streams subject to this stress gradually become eroded and the water table becomes increasingly deep (Sweeney et al., 2004). The riparian zone gradually loses the water saturation gradient that supports a diversity of wetland species. If agricultural production stops, the riparian zone will not necessarily be recolonized by species characteristic of floodplains since the biophysical connectivity between the vegetation, water table and stream has been reduced or eliminated. In such a context, the restoration of the riparian zone should not be focused on original vegetation types (Dosskey *et al.*, 2010), but rather on species adapted to the new conditions. In Quebec, the hydrology and morphology of agricultural streams are often altered by drainage, bank straightening and regulation (Beaulieu, 2001). Their banks do not always support the same assemblage of plant species that characterizes the riparian zones of natural environments.

Agricultural activities (tillage, fertilization, drainage, liming) also alter soil properties and often cause homogenization, sometimes to such an extent that the composition of 100-year-old secondary forests (post-agricultural) is altered relative to primary forests (Flinn and Marks, 2007; Vellend *et al.*, 2007). However, an analysis of the functional traits of the plant species composition of the Quebec sugar maple forests revealed few differences between unmanaged old-growth forests and regenerated forests that had been cleared in the 19th century for agriculture (Aubin *et al.*, 2007). The authors conclude that the understory vegetation of the Laurentian forest can maintain its functionality despite major disturbances and that the overstory of the forests studied is able to naturally regenerate in the long term. The ability of tree species to colonize certain riparian zones is questionable due to heavy competition by herbaceous vegetation and the absence of seed trees in the landscape.

The significant connectivity that exists between linear wetlands (e.g., road ditches and agricultural drainage ditches) can also facilitate invasion of agricultural riparian corridors by aggressive exotic species, such as common reed (*Phragmites australis* (Cav.) Trin. ex Steud; Figure 23) (Maheu-Giroux and de Blois, 2007). The full light conditions that prevail in various deforested agricultural riparian areas facilitate colonization by exotic plants, since most of them are shade intolerant (Parendes and Jones, 2000; Martin *et al.*, 2009).

A number of measures can be considered for restoring native communities affected by the invasion of exotic plant species: restoring the hydrologic regime, stabilization of stream banks, direct control of exotic species by biological or mechanical control, managing the disturbance regime (fire, grazing) and establishment of competitive native plants (Richardson *et al.*, 2007). These measures can be used alone or in combination, depending on the context and the plant community to be restored.

In what were once forest landscapes in Wisconsin, forested riparian zones in agricultural landscapes supported more native plant species than riparian zones dominated by grasses or grazed by cattle (Paine and Ribic, 2002). In the agricultural watershed of the Boyer River, ephemeral spring plants grow primarily in forested riparian habitats (Boutin *et al.*, 2003); they are indicator species of the ecological integrity of the Laurentian forest (Boutin *et al.*, 2003; Aubin *et al.*, 2007). However, no species of conservation interest were surveyed in the study by Boutin *et al.* (2003), even in wooded riparian habitats.

Figure 24: **Log jam formed by dead trees (left). A pileated woodpecker feeding on a snag (right). The photographs were taken in a riparian zone in Parc du Domaine Vert, Mirabel (Quebec) (photo credit: J. Fortier).**

4.2.4. Dead wood

As described in the table in Appendix 2, dead wood, particularly coarse dead wood, is an important component of riparian ecosystems, both for water quality and for terrestrial and aquatic habitat quality (Fetherston *et al.*, 1995; Abbe and Montgomery, 1996). Coarse woody debris on the ground and in streams can also significantly affect the hydrologic regime of streams (Tabacchi et al., 2000). Dead wood has important functions in all of its forms: snags, fallen trees, stumps, log jams, submerged wood or beaver dams (Figure 24). When combined with other indicators, the complexity of dead wood within riparian zones can also be used to characterize the health of riparian and aquatic habitat (Quinn *et al.*, 1997a).

4.3. Spatial heterogeneity

At the landscape scale, agricultural riparian zones are unique corridors within the agricultural matrix. They act as conduits for movements of matter and species across landscapes, but also as barriers to perpendicular flows. It is possible to differentiate between the various patches that compose the riparian mosaic on the basis of vegetation cover, which is dependent on the water saturation conditions of the soil, natural and human disturbances and the hydrologic dynamics (Figure 25) (Naiman *et al.*, 2005). Each patch has a particular assemblage of species, a distinct hydrogeological context and a specific biophysical connectivity.

As a result, at the landscape scale, there exist dynamic and complex interactions not only among riparian patches, but also between patches and the agricultural matrix. As a result, the various vegetation patches produce specific ecosystem services in the agroecosystem (Paine and Ribic, 2002; Boutin *et al*., 2003; Jobin *et al.*, 2004; Naiman *et al*., 2005; Knight et al., 2010). Moreover, a riparian corridor that is complex and heterogeneous from the point of view of its vegetation cover may indicate that the hydrologic complexity of the corridor has been maintained.

The restoration and local protection of natural habitats appears to be more important in homogeneous landscapes dominated by intensive agriculture than in already complex agricultural landscapes (Andrén, 1994; Tscharntke *et al*., 2005). In other words, the potential gains in terms of habitat quality and overall diversity are higher if intensively managed homogeneous landscapes are restored (Tscharntke *et al*., 2005).

4.4. Stresses on agricultural riparian zones

As we saw in section 3.1, the concept of stress is associated with the concept of the health of a natural system (Rapport *et al*., 1998). For any system, the level of stress to which it is subject will depend on the nature of the disturbances, but also on disturbances that affect the surrounding landscape. This means that two riparian zones with similar plant species composition could have a different state of health due to the fact that they are not affected by the same stresses or to a comparable level of stress (e.g., forested riparian buffer along a corn field vs along a fenced pasture).

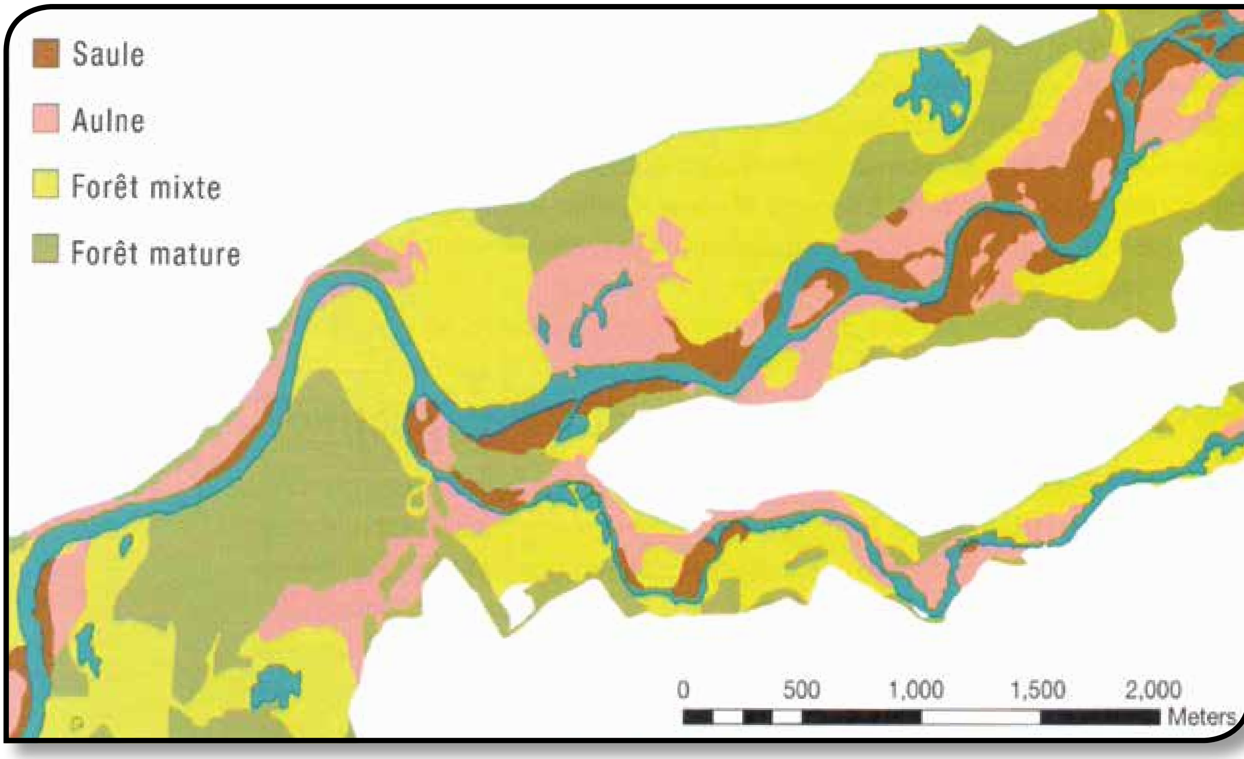

Figure 25: **Along the riparian corridor of the Queets River (Washington), the migrating river channel is responsible for the establishment of various species. The patches are identified on the basis of the dominant plant species: willow, alder, mixed spruce forests and mature spruce forest (Naiman *et al.*, 2005).**

Unlike undisturbed riparian ecotones, which are only subject to natural disturbance and stress (floods, windfall, forest fires, insect outbreaks, grazing, etc.), agricultural riparian zones are also exposed to many anthropogenic stresses that sometimes magnify the effects of natural disturbances. It is therefore important to clearly characterize the agricultural environment surrounding the riparian zone in order to identify all sources of stress that can compromise the development and sustainability of the riparian system.

Resilience to a given stress differs depending on the type of system affected by the stress, although a number of general trends are observed (Appendix I) (Odum, 1985; Groffman *et al.*, 2006), Several effects are observed depending on fluctuations in the intensity of the stress in time (i.e., either delayed, acute or chronic, and either increasing, stable or decreasing). In addition, new stresses that can appear suddenly can exacerbate existing stresses.

For these reasons, ecosystem responses to stress are often non-linear and diffuse in time (Groffman *et al.*, 2006). It is difficult to predict the complete response of a system to stress, particularly when numerous stresses are involved (Rapport, 1992).

From a riparian zone health assessment perspective, it is important to establish the extent of the stresses present (Figure 26) (Stoddard *et al.*, 2005). It is also important to characterize their frequency, persistence, severity and spatial-temporal complexity (Stoddard *et al.*, 2005) and to rank the

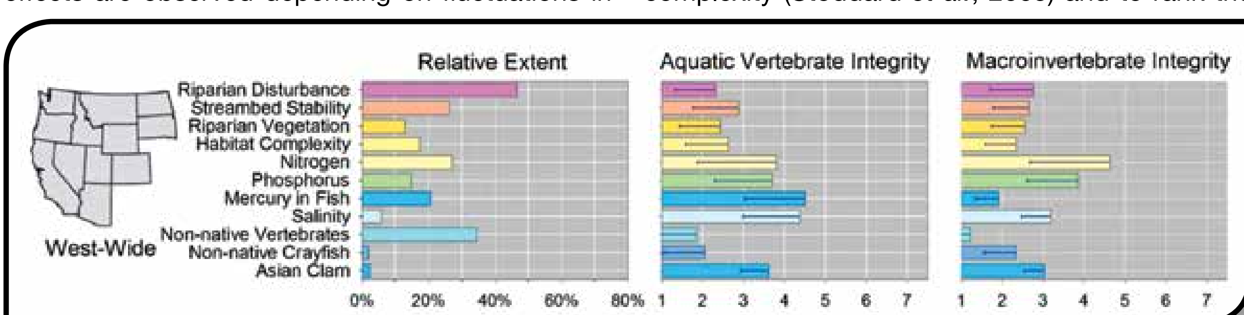

Figure 26: **Relative extent of disturbances per km of stream, and relative risk posed by each disturbance to aquatic vertebrate integrity and macroinvertebrate integrity in small streams and rivers of the Western United States (West-Wide) (Stoddard *et al.*, 2005).**

various stresses to which agricultural riparian zones can be exposed in order of importance.

This section contains a brief overview of a number of natural and human-induced stresses that could compromise the development and organization of a riparian system.

4.4.1. Human-induced stresses

Tree clearing

Clear-cutting of woody vegetation in riparian zones is a major stress because it moves the systems back to a less advanced state of development, i.e., to early successional stages (Figure 26). Complete clearing of riparian zones can have a number of effects on riparian habitat:

- reduction of the capacity to intercept light on stream edges and to create a microclimate (effect on the thermal regime of the water and growth of algae);
- loss of forest habitat; and
- increased vulnerability of the system to shade intolerant invasive plants.

The impacts associated with partial clearing of woody vegetation will likely be less severe, because the existing tree structure is conserved.

Conversion to agricultural use

Clearing of riparian zones followed by conversion to cropland or pasture is a much more significant stress in terms of severity and duration than clearing followed by the regeneration of natural vegetation. An agricultural riparian zone that is subjected to this type of stress generally sees it trajectory deviate from its original trajectory (Figure 9). When a natural riparian zone is converted to annual crops or pasture, the following general trends may be observed (Kauffman and Krueger, 1984; Quinn *et al.*, 1992; Quinn *et al.*, 1997a; Paine and Ribic, 2002; Sweeney *et al.*, 2004; Flinn and Marks, 2007; Dosskey *et al.*, 2010):

- Increasingly fragile stream banks
- Erosion of bare soil due to tillage (when applicable)
- Disturbance of the nature of soil (organic matter content, spatial heterogeneity, porosity, compaction, etc.) associated with tillage and livestock trampling
- Replacement of natural vegetation cover and loss of riparian habitats
- In pastures, decline in riparian plant vigour and biomass due to livestock browsing
- Lowering of the water table
- Vertical stream erosion
- Lower retention/transformation of pollutants
- Decline in water infiltration
- Decline in aquatic habitat protection
- Loss of microclimate
- Increase in chemical inputs (fertilizers, pesticides, amendments)
- Invasion by exotic species promoted by habitat degradation

Agricultural activities adjacent to riparian zones

A natural riparian buffer strip bordered by an annual crop is affected by certain stresses associated with agricultural practices carried out in the adjacent field. For example, aerial drift of herbicides sprayed in the field could compromise the vigour and composition of riparian communities and even the survival of certain wetland plant species (Boutin and Jobin, 1998; Carpenter and Boutin, 2010; Dalton and Boutin, 2010).

Fertilizers that are used in annual crops cultures can migrate in riparian soil and adversely affect certain functions of riparian vegetation. Sabater *et al.* (2003) report that the potential for reduction of nonpoint source nitrogen pollution is negatively correlated with the nitrate load that reaches the riparian zone. When present in the soil at excessive concentrations, ammonium and nitrate can bet toxic to some plants (van der Eerden, 1982). Conversely, nitrate enrichment of soil favours nitrophiles, including a number of introduced plant and weed species, to the detriment of native plant species (Reinhart and Callaway, 2006).

Annual crops can also generate sediment that is deposited in the riparian zone (Knight et al., 2010), thereby creating sites conducive to the establishment of opportunistic species, including various invasive alien plants (Richardson *et al.*, 2007).

Alteration of hydrology

The agricultural riparian zone is defined on the basis of soil moisture gradient (Nilsson and Berggren, 2000; Naiman *et al.*, 2005) and is therefore sensitive to hydrologic alterations. Land drainage lowers the level of the water table, drying out the soil and making it more likely to support upland and invasive plant species rather than wetland, often native plant species (Richardson *et al.*, 2007).

Various functions associated with the biophysical connectivity between vegetation, the water table and the stream are adversely affected by drainage (see section 4.1). In watersheds where agricultural drainage is extensive, high, short-duration peak flows are observed due to the lack of water retention in the agroecosystem (Government of Quebec, 2007). These high flows contribute to processes such as stream bank and bed erosion.

In the Prairies, the construction of dams for irrigation purposes is a significant cause of downstream degradation of riparian poplar forests (Rood and Mahoney, 1990; Cordes *et al.*, 1997). Historically very abundant, these riparian forests require a natural hydrologic regime for the establishment and development of young trees (Rood and Mahoney, 1990). By restoring a more natural hydrologic regime, i.e., flooding and drying cycles, it is possible to restore fish and aquatic bird communities (Toth et al., 1998). A conceptual diagram of the main ecological impacts of flow regulation is presented in Figure 27.

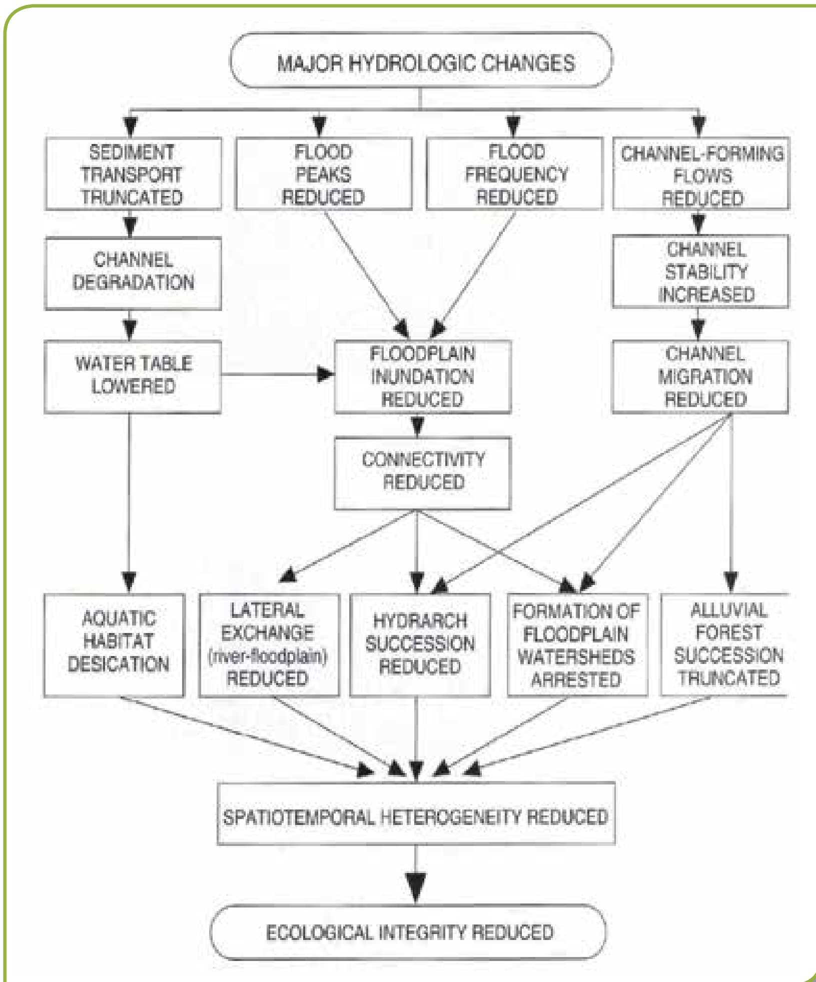

Figure 27: **Conceptual diagram of the potential ecological impacts of flow regulation (from Ward and Stanford, 1995).**

Alteration of stream morphology

The straightening of drainage and stream channels to promote rapid drainage alters the hydrologic regime, bank structure and vegetation. Like drainage, this measure lowers the water table.

Alteration

The alteration of the riparian zone by the construction of infrastructure (retaining walls, roads, bridges, homes, business, etc.) leads to the replacement of banks and riparian vegetation with materials whose sole function is to withstand erosion. Retaining walls, which are often used along urban streams, prevent hydrologic and biophysical exchanges between the water table and the stream (Kaushal *et al.*, 2008). Filling eliminates the riparian zone and wetlands, which causes the degradation of habitat and functions associated with the riparian zone.

4.4.2. Natural stressors and disturbances

Flooding and drought

The flooding regime has a significant impact on the survival and distribution of riparian plant species. Flooding is a natural disturbance characteristic of riparian zones. It is generally conducive to the colonization of the area by tolerant species. Extreme flooding events can cause the decline of flood-intolerant species that colonize the riparian zone (Vervuren *et al.*, 2003). They can also favour the invasion of the riparian zone by exotic plants (Hood and Naiman, 2000). Conversely, prolonged drought can lead to the decline of obligate aquatic plants and promote the introduction of facultative wetland plants (Smith *et al.,* 1998).

Climate change will likely have significant effects on the natural disturbance regime in riparian zones, particularly in terms of flood and drought recurrence intervals (Lehner *et al.*, 2006).

Ungulate grazing

A large population of ungulates can be a significant stressor for riparian vegetation. Deer regularly feed on young woody stems and can adversely affect natural regeneration or the establishment of plantations (Opperman and Merenlender, 2000; Ripple and Beschta, 2003; Sweeney and Czapka, 2004).

Invasive alien plants

Invasive alien plants disturb native riparian communities and biodiversity in general. Biological invasion is considered a factor (or general stressor) that affects the functions, structures and assemblages of species in an ecosystem (Vitousek *et al.*, 1996).

5. LINKAGES BETWEEN AGRICULTURAL RIPARIAN HEALTH AND AQUATIC HABITAT QUALITY

5.1. Is land use in a watershed a better indicator of aquatic habitat quality than riparian zone health?

Aquatic habitat quality depends on factors that have an effect at various spatial scales. Studies conducted at several scales provide valuable insights into the linkages that may exist between aquatic habitat quality and land use. These multi-scale studies generally assess land use at three separate scales: (1) riparian buffer between 30 m and several hundred metres wide on each bank for stream segments varying in length from 100 m to approximately 1 km; (2) riparian buffers of the same width, but for the entire section of the stream located upstream of the sample site (reach buffers) and (3) the entire catchment located above the site (Figure 28) (Allan, 2004).

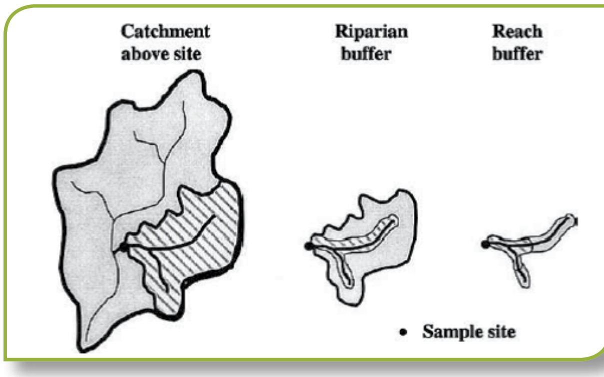

Figure 28: The three spatial scales most commonly used in studies of the linkages between landscape variables and the physical and biological characteristics of the stream: catchments above site, riparian buffers located upstream of the sample site and buffers located in the immediate vicinity (reach buffers) (Allan, 2004).

Some environmental variables vary according to local and regional factors (Figure 29 and Figure 30) (Frissell *et al.,* 1986). For example, shade is a function of the local composition of riparian vegetation, which affects water temperature over distances of sometimes over 1 km (Quinn *et al.*, 1992; Quinn *et al.,* 1997b). Local factors such as riparian vegetation and stream morphology also affect inputs and retention of allochthonous organic material (leaves and wood) as well as aquatic habitat structure (Allan *et al.*, 1997; Oelbermann and Gordon, 2001).

However, it is often land use within the entire watershed that influences the state of health of a stream. Studies that have examined the effect of agricultural land use on aquatic habitat quality at various spatial scales have difficulty showing that local factors, in terms of vegetation cover or occupation of the riparian zone, have the greatest influence. For example, land use has an influence on hydrology, evapotranspiration, and infiltration and runoff, and is therefore a significant indicator of nutrient loading to streams (Boyer *et al.*, 2002; Allan, 2004).

Roth *et al.* (1996) and Allan *et al.* (1997) have determined that riparian vegetation at the local scale was a secondary and weak indicator of the integrity of the aquatic environment of the agricultural stream studied, and that the width of the riparian zone occupied by tree vegetation did not appear to have a significant effect on aquatic habitat quality. The results suggest that the index of biotic integrity and the habitat index are negatively correlated with the area occupied by agricultural land and positively correlated with the area occupied by wetlands and forests. These correlations were stronger at the regional subwatershed scale and became weaker and not significant at the local scale (Table 3).

Wang *et al.* (1997) analyzed the relationships between watershed land use and aquatic habitat quality at 134 sites located on 103 streams in Wisconsin. Habitat quality and index of biotic integrity were positively correlated with the amount of forested land and negatively correlated with the amount of agricultural land in the subwatershed and riparian zone (100-m wide buffer along the stream). Correlations were generally stronger for the subwatershed than for the riparian zone (buffer).

Stephenson and Morin (2009) observed that forest cover at the watershed scale provided a better explanation of plant community structure and biomass of algae, invertebrates and fish than plant

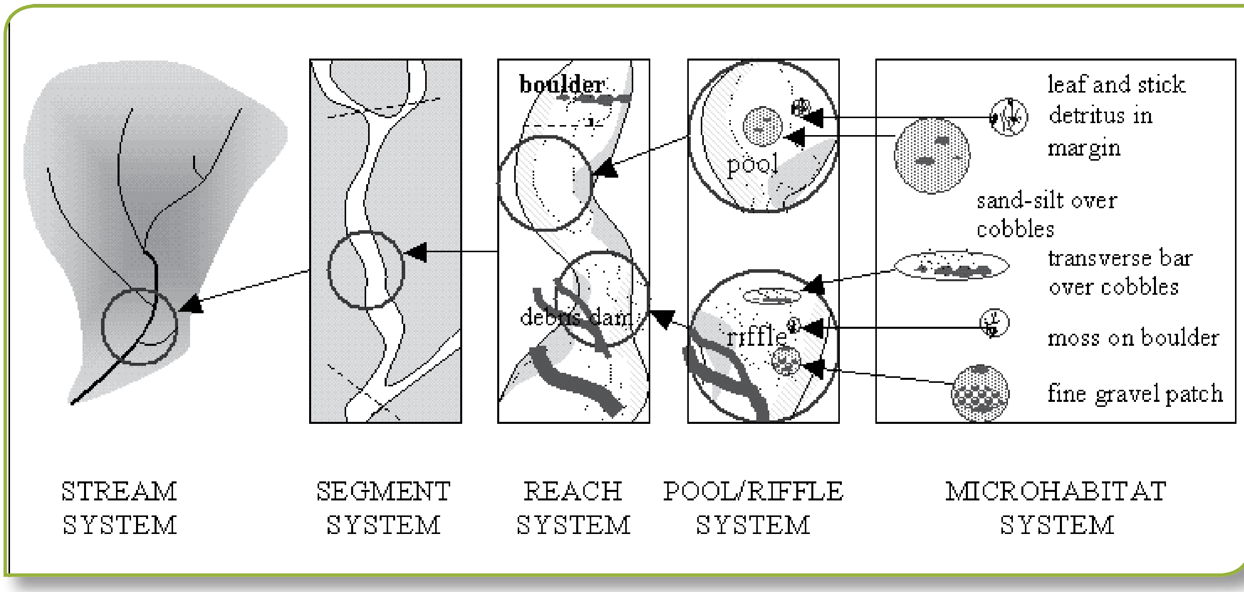

Figure 29: **Landscape influences on stream ecosystem structure and function across spatial scale. The diagram shows the hierarchical relationships among habitat and landscape features of streams (Frissell et al., 1986), from the smallest to the largest: boulder cascade, debris dam, pool/ riffle, leaf and stick detritus in margin, sand/silt over cobbles, transverse bar over cobbles, moss on boulder, fine gravel patch.**

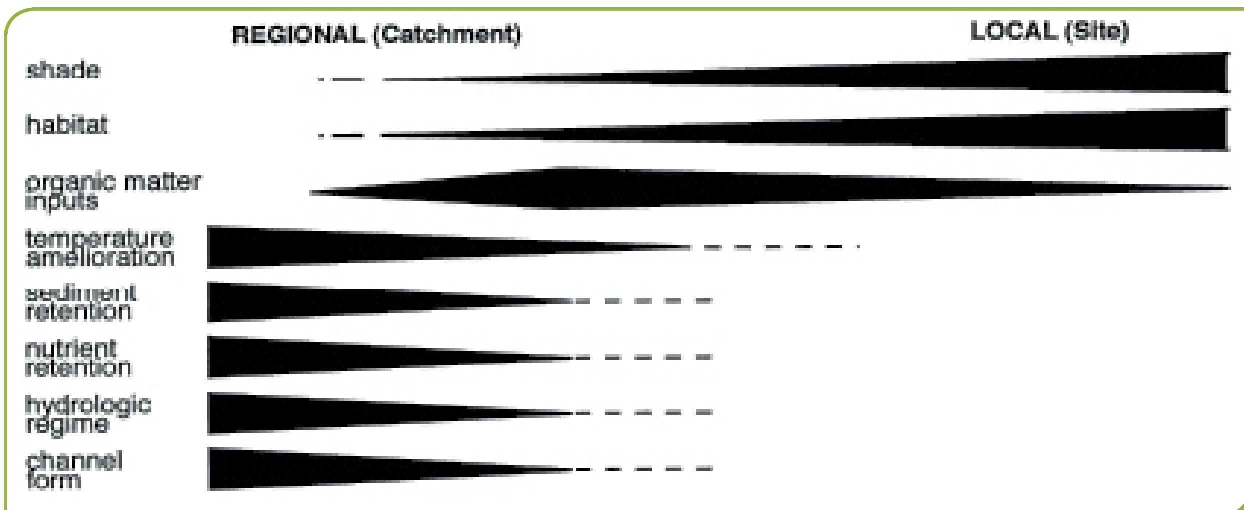

Figure 30: **Speculative account of the influence exerted by local (10 to 1000 m) versus regional (over 1 km) terrestrial vegetation over stream function (Allan *et al.*, 1997).**

Table 3: **Relationship between the proportion of cultivated area at various spatial scales and the habitat index and the index of biotic integrity measured on fish communities (Roth *et al.*, 1996; Allan *et al.*, 1997). Regression coefficients marked with an * are significant at p<0.05.**

		150 m section	1.5 km segment	Total stream length	Watershed
Habitat index	R^2	0,164	0,160	0,533*	0,758*
Biotic integrity index	R^2	0,017	0,073	0,378*	0,496*

cover at the local and regional scale. The study was conducted in the Ottawa region (Quebec and Ontario), in 38 riparian forests at 55 different spatial scales of forest cover (riparian forest over widths of 10 to 320 m and lengths of 10 to 1280 m, entire riparian distance upstream from the sampling areas).

Another study conducted on 25 agricultural streams in Wisconsin showed how difficult it is to isolate the relative influence of natural and anthropogenic factors on aquatic habitat quality given that these factors tend to co-vary (Fitzpatrick *et al.*, 2001). That study reported that the effect of environmental characteristics varied depending on the biotic indicator used. For example, watershed and buffer land cover (50 m in width), geomorphology, reach riparian vegetation width, and stream size all had an effect on the index of biological integrity calculated for fish and diatoms, as well as on invertebrate diversity and the number of algal taxa. However, the percentage of EPT (Ephemeroptera, Plecoptera and Trichoptera), the benthic invertebrate family biotic index and the diatom pollution index were better correlated with nutrient concentrations and flow variability. In contrast, fish IBI scores were more sensitive to land cover in the entire stream network buffer.

A number of studies also show that local factors, in terms of riparian vegetation cover, are more strongly correlated to integrity indices in benthic invertebrate and fish communities. According to observations made in three agricultural subcatchments of the River Raisin watershed in Michigan, Lammert and Allan (1999) observed stronger correlations between aquatic variables and vegetation cover type in a 100-m wide riparian zone compared to vegetation cover at the watershed scale. Although significant, the correlations between the indices of biotic integrity and riparian vegetation explained only a modest amount of the variability observed. The strongest correlation was between the percentage of riparian forest (100 m wide) and the benthic invertebrate index of biotic integrity (r=0.35). The models obtained using multiple linear regression produced stronger correlations, with the best models containing variables characterizing local habitat (percentage of riparian forest, flow stability, substrate type, stream width).

The work of Sponseller *et al.* (2001), carried out in nine Appalachian basins characterized by a mixture of land use practices, also suggest that vegetation cover at a local scale (30 m in width and covering 200 m of stream) is more strongly correlated to benthic indices than when it is measured at larger scales. The authors believe that the beneficial effect of riparian forest patches on the thermal regime is critical to the distribution of various species in streams draining catchments with mixed land use practices.

A study of the relationships between vegetation cover and benthic communities at three spatial scales in 33 sub-basins of the Upper Thames River in Ontario suggests that the proportion of forest cover and agricultural land cover at the local scale influences benthic communities, whereas no significant relationship was observed at the subbasin scale (Rios and Bailey, 2006).

A study conducted in Michigan on three watersheds characterized by mixed land uses (forests, agriculture, city, industry) suggests that macroinvertebrate community structure and functions are more highly influenced by local-scale habitat factors than by land use at the watershed scale (Stewart *et al.*, 2000).

Other studies suggest that both local factors and watershed factors have an influence on the composition of stream macroinvertebrate communities. That is the case of a study comparing streams adjacent to cornfields, pasture with unrestricted access to streams, and forest sites (Kyriakeas and Watzin, 2006).

In Quebec, the riparian strip quality index (IQBR) evaluated over a width of 30 m along the Chaudière River, a higher order stream, was not significantly correlated with the index of biotic integrity (r=0.23; p=0.22), calculated with fish communities, or the overall biotic index (r=0.33; p=0.07), calculated with invertebrate communities (St-Jacques and Richard, 1998). However, a stronger correlation could have been obtained between the IQBR and the biotic indices if the methodology had been applied to small streams, where the type of riparian vegetation has a greater influence on aquatic habitat quality.

5.2. Stream biodiversity: the ghost of land use past

The work of Harding *et al.* (1998) supports the hypothesis that the influence of past watershed land use (50 years ago) on present invertebrate and fish diversity in streams is greater than that of the presence of forests in riparian zones. The authors sample benthic invertebrate and fish communities on 24 streams (12 in agricultural watersheds and 12 in forested watersheds) in two river basins in North Carolina to calculate different indices of ecological integrity and aquatic habitat quality. These indices were then correlated to the percentage of the watershed in forest at two temporal scales (1950 and 1990) and seven spatial scales. Based on the results obtained, the authors suggest that large-scale and long-term agricultural disturbances in a watershed limit the recovery of stream diversity for several decades. Moreover, although the proportion of riparian forests within a 30-m riparian zone had increased by 30% from 1950 to 1990 in one of the watersheds, this does not appear to have had a significant effect on aquatic wildlife when compared to streams in forested areas. This suggests that the process of recovery of aquatic communities takes decades. Past practices in terms of whole watershed land use can explain the current status of aquatic biodiversity.

5.3. Effect of channelization, drainage and vertical erosion on aquatic habitat

As mentioned a number of times, channelization of streams in agricultural areas for the purpose of drainage has major impacts both on the riparian zone and on aquatic habitat. A study conducted in southern Ontario shows that channelization and subsurface tile drainage practices eliminate most of the benefits associated with natural riparian buffers and magnify the effects of adjacent farming practices on benthic fauna (Barton, 1996). Osborne and Kovacic (1993) also observed that the positive effects of riparian buffer strips on water quality are significantly reduced in the presence of a subsurface drainage. These results may explain why Barton (1996) did not observe any relationships between the health of invertebrate communities and the quality of the riparian buffer of channelized and drained streams. On natural streams, a very weak correlation was observed between the quality of the riparian buffer and the health of invertebrate communities, thereby providing additional support for the hypothesis that cover in the watershed may be the best indicator of aquatic habitat quality (Barton, 1996; Roth *et al.*, 1996; Allan *et al.*, 1997; Harding *et al.,* 1998; Fitzpatrick *et al.,* 2001; Stephenson and Morin, 2009).

Natural vertical erosion of streams in agricultural areas is a significant cause of the loss of aquatic habitat quality. As described by Quinn *et al.* (1992), vertical stream erosion reduces the shade provided by riparian vegetation. Maximum daily water temperatures during the summer are therefore higher, which has an impact on invertebrate communities. The authors suggest that the factors affecting the aquatic habitat of grazed streams less than 6 m in width are, in order of importance: (1) vertical stream erosion; (2) intensive grazing of riparian vegetation; and (3) extensive grazing of riparian vegetation by livestock.

Sweeney *et al.* (2004) observed that deforested agricultural streams incised by erosion provide less habitat for fish and invertebrates, and have less potential for the assimilation of ammonium, phosphorus and pesticides.

5.4. Effect of riparian vegetation on stream thermal regime

In contrast to the conflicting relationships that exist between riparian buffer quality and aquatic habitat quality, the presence of forested buffers in agricultural areas is better correlated with mean and maximum daily water temperature, particularly on small streams (Quinn *et al.*, 1997a; Lyons *et al*., 2000; Rutherford *et al.*, 2004; Caissie, 2006). The thermal regime is influenced primarily by direct solar radiation which, in turn, is directly influenced by the type of vegetation cover in riparian zones (Johnson and Jones, 2000).

Given that living organisms have water temperature preferences, the thermal regime is known to have an important effect on a number of organisms, ranging from invertebrates to salmonids (Caissie, 2006).

5.5. Effect of riparian vegetation on algal growth

A number of studies suggest that canopy opening (or incident radiation) influences the development of algae in streams. In an experiment in a semi-controlled environment, Quinn *et al*. (1997b) observed that algal blooms are virtually non-existent with canopy opening of between 2% and 10%, relatively rare with canopy opening of 40%, and frequent with canopy opening of 100%. Streams draining native forest can have up to 30-fold lower algal biomass than streams draining pasture (Quinn *et al.,* 1997a). Munn et al. (2010) also observed that open canopy streams (i.e., canopy cover of less than 50%) had higher total algal biomass.

Scenarios tested by Ghermandi *et al*. (2009) in a simulation indicate that it is possible to control eutrophication on streams of low to moderate width by reducing phytoplankton productivity by 44% by creating shade (Ghermandi *et al.*, 2009). According to these authors, shade has a limited effect on other important variables, such as dissolved oxygen, chemical oxygen demand and nutrient concentration. Nutrient concentration, water temperature and water velocity are also known to have a significant influence on algal growth in agricultural streams (Munn *et al.*, 2010).

6. ASSESSING AGRICULTURAL RIPARIAN HEALTH

6.1. How to assess the health of a system

The assessment of ecosystem health always begins with the premise that the natural system under study is imbalanced or degraded and that a diagnosis is required (Figure 31) (Jorgensen, 2010). Once the imbalance is observed, specific indicators are selected and used to conduct a health assessment. These indicators should make it possible to identify the main causes of the degradation of the system under study. Once the health assessment is completed, a precise diagnosis should be presented and should serve as a basis for the development of a response plan that will lead to an improvement in the health of the system. Other indicators can be used to monitor the state of health of the system following the action taken.

Although there are general ecological indicators that are commonly used to provide a health diagnosis, it is often necessary to identify indicators specific to the system being studied (Costanza *et al.*, 1992) on the basis of the user's needs. There is therefore no procedure for selecting ecological indicators.

6.2. Characteristics of a good health indicator

Costanza *et al.* (1992) suggests that a good health indicator (or good series of indicators) should be able to take the organization and resilience of a system into account in a specific environmental context. To do so, it is essential to select indicators that are able to clearly identify the structure and functions of the system.

From a practical environmental management point of view, a good health indicator should be (Jorgensen et al., 2010):

1) simple to apply and easily understood by land use managers;
2) relevant to the specific context of the system being studied;
3) scientifically justifiable;
4) quantitative;
5) acceptable in terms of cost.

From a scientific point of view, a good ecological indicator should be (Jorgensen *et al.*, 2010):

1) easily measurable;
2) sensitive to small variations in environmental stress;
3) independent of a reference state;

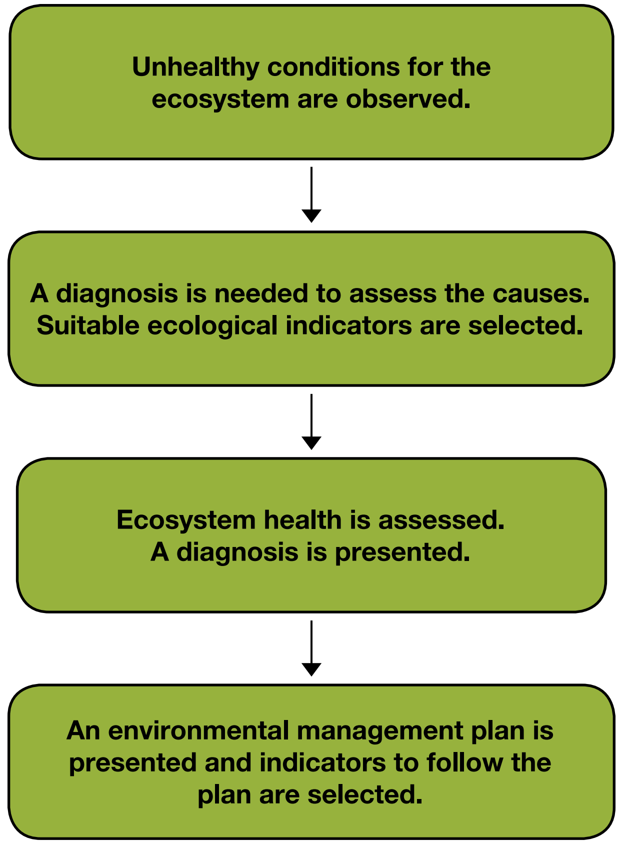

Figure 31: Use of ecological indicators to assess ecosystem health and to monitor the effect of an environmental management plan (Jorgensen, 2010).

4) applicable in extensive geographic regions and to a large number of communities;
5) quantitative.

6.3. Categories of ecological indicators

Indicators used to measure the health of ecosystems or natural systems can be classified from the most reductionistic to the most holistic (Jorgensen *et al.,* 2010):

1) Presence or absence of specific species (e.g., the determination of the trophic status of a lake was based on the presence/absence of specific fish species).
2) Ratio between particular species (e.g., there are several indices calculated with invertebrate and fish communities that are used to assess various aspects of stream health).
3) Concentrations of chemical compounds (e.g., total nitrogen in water is often used as an indicator of water and habitat quality).
4) Measurements of biomass or density of entire trophic levels (e.g., concentration of phytoplankton, fish, invertebrates, birds).
5) Rates for specific processes, such as primary production or respiration.

6) Composite indicators which integrate several other indicators in terms of energy, structure, evolution and homeostasis (e.g., the use of the ratio respiration/primary production, primary production/biomass, primary producer/consumer). Composite indicators can be used to assess whether an ecosystem is at an early stage of development or is a mature ecosystem (Odum, 1969).
7) Holistic indicators that take account of several functions and underlying structures, such as resilience, buffer capacity, all forms of biodiversity, size of ecological networks and food chains.
8) Thermodynamic indicators, which are used to assess variables such as energy stored in the system (energy), capacity of a system to perform work (exergy), disorder (entropy), etc.

The various indicators selected can then be measured and combined to create a health indicator using various approaches (Jorgensen *et al.*, 2010):

1) *Direct measurement method.* Once the indicators are selected, they are measured directly in the field or calculated indirectly from field observations. Ecosystem health is assessed on the basis of the resulting indicator values.
2) *Ecological modeling method.* For this approach, the model structure is determined on the basis of the structure and complexity of the system studied. This makes it possible to design a conceptual diagram, develop model equations and estimate model parameters. The model is then calibrated in order to calculate ecosystem health indicators. Ecosystem health is then assessed on the basis of the values of the indicators.
3) *Ecosystem health index method.* The ecosystem obtains a score between 0% and 100%, depending on its state of health, which can range from «bad» to «excellent». To facilitate the interpretation of the index, EHI is divided into five segments or ranges as follows: 0-20% = worst, 20-40% = bad, 40-60% = middle, 60-80% = good and 80-100% = excellent. The index can be calculated by the following equation:

$$ISE = \sum_{i=1}^{n} w_i * ISE_i$$

where EHI is a synthetic ecosystem health index containing all of the health indicators selected (sub-EHIs), EHI_i is the *i*th ecosystem health index for the ith indicator and w_i is the weighting factor for the *i*th indicator. The procedure for developing this type of index is illustrated in Figure 32.

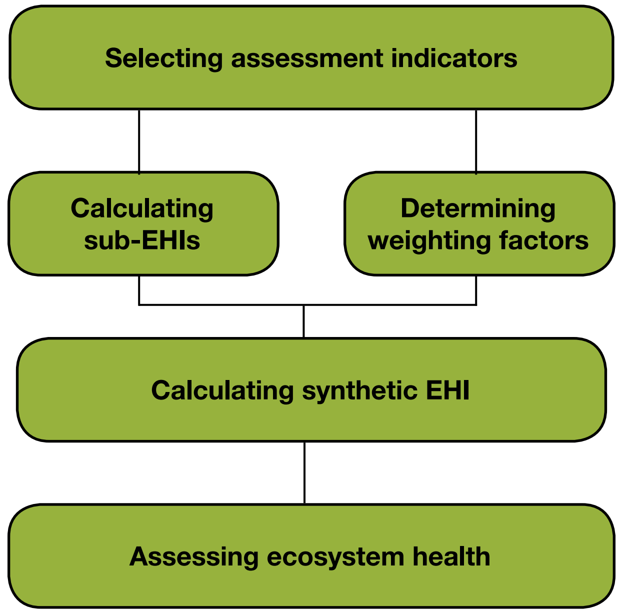

Figure 32: Procedure used to calculate an ecosystem health index integrating several weighted sub-EHIs (Jorgensen *et al.*, 2010).

6.4. Potential indicators of agricultural riparian health

6.4.1. Indicators of biophysical connectivity

Biophysical connectivity between riparian vegetation, the stream and groundwater is a key factor in determining the capacity of an agricultural riparian zone to protect water and terrestrial and aquatic habitat quality (section 4.1). Indicators that can be used to assess the state of biophysical connectivity include the following:

Extent of the riparian zone

The extent of the riparian zone can be identified using light detection and ranging (LiDAR) data, as shown by Johansen *et al.* (2010), and is an indicator that can be used to identify the riparian zone (Figure 33), i.e., the zone where there is potentially a significant interaction between vegetation, the water table and the stream. As a result, the larger the riparian zone, the greater the interaction between the vegetation and the groundwater and the more it can be colonized by wetland plants and used by wetland wildlife (amphibians, waterbirds, martens, beavers, etc.). The extent of the riparian zone could

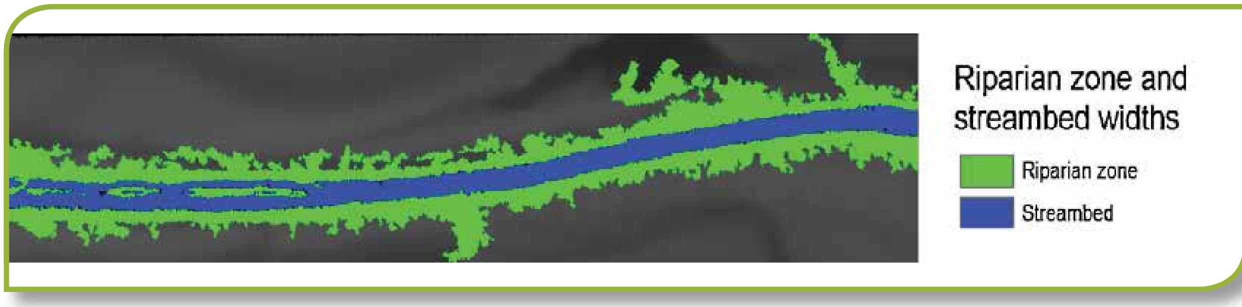

Figure 33: Identification of the riparian zone using LiDAR data (Johansen *et al.*, 2010).

therefore be an indicator equivalent to the wetness index recently proposed by Tomer *et al.* (2009). To quantify the extent of the riparian zone, one could simply calculate the area occupied by the riparian zone for each stream reach. The average width of the riparian zone could also be quantified for each stream reach.

Vertical stream erosion

The degree of vertical stream erosion can be calculated by measuring the vertical distance between the stream during low flow conditions and the top of the bank. It identifies riparian zones that still have a water saturation gradient, as well as those that have lost this gradient and whose connectivity with the stream has been altered. This information was taken from digital elevation models obtained using LiDAR data (Hall *et al*., 2009).

Density of the stream drainage network and abundance of intermittent streams

Given that the effectiveness of riparian zones in protecting aquatic habitat and water quality is substantially reduced when they are bypassed by small intermittent agricultural streams (Osborne and Kovacic, 1993), it is important to develop an indicator that can quantify the extent of the stream drainage network of the various reaches of streams. A hydrographic map, when not available for the sector in question, can be developed using satellite images or a digital elevation model obtained by LiDAR (Figure 34). Other preferential flow paths could also be identified using LiDAR.

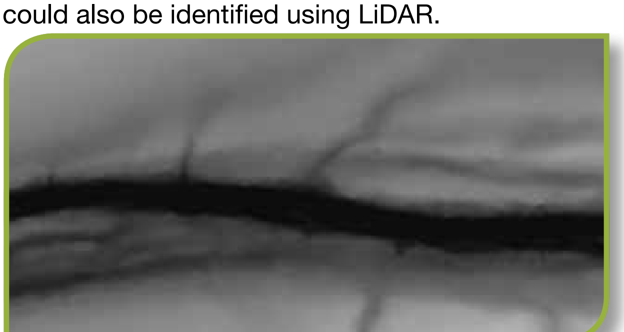

Figure 34: On this digital terrain model obtained using LiDAR data, it is possible to identify small streams or wetland depressions that are intermittent tributaries of the main stream (Johansen *et al.*, 2010).

Longitudinal connectivity (continuity)

This indicator evaluates areas where there are canopy gaps in a riparian forest corridor. This indicator is important for terrestrial wildlife, since certain species will be reluctant to use a corridor if it does not offer continuous or relatively unfragmented forest cover (Bentrup, 2008). It can also be calculated using LiDAR models (Figure 35). Examples of the calculation of this indicator could be (1) the number of gaps per reach of stream;(2) the average area of canopy gaps per reach of stream; and (3) the total area of canopy gaps per reach of stream.

6.4.2. Indicator of vegetation structure and composition

Vegetation structure and composition has an important influence on several riparian processes, as well as on habitat quality (Appendix 2). There are a number of indicators that can be used to quantify vegetation structure and composition:

Plant cover height

Cover height provides indications of the dominant vegetation strata. It also provides a relatively precise indication of the spatial arrangement of vegetation patches. By means of a canopy elevation model obtained using LiDAR data, it is possible to create various categories of cover height in order to identify herb, shrub and tree layers (Figure 36) (Johansen *et al.*, 2010). The percentage of area occupied by each layer per reach of stream can then be calculated on the basis of these categories.

This approach can be used to identify zones where woody vegetation is becoming established among herbaceous vegetation. This is an important indicator of health since the colonization of a riparian buffer by woody vegetation is a process that will enable the riparian zone to evolve to a more complex and more functional organizational state. A riparian zone denuded of trees is not necessarily in a poor state of health, since soil water saturation conditions are sometimes responsible for the lack of trees (Figure 22).

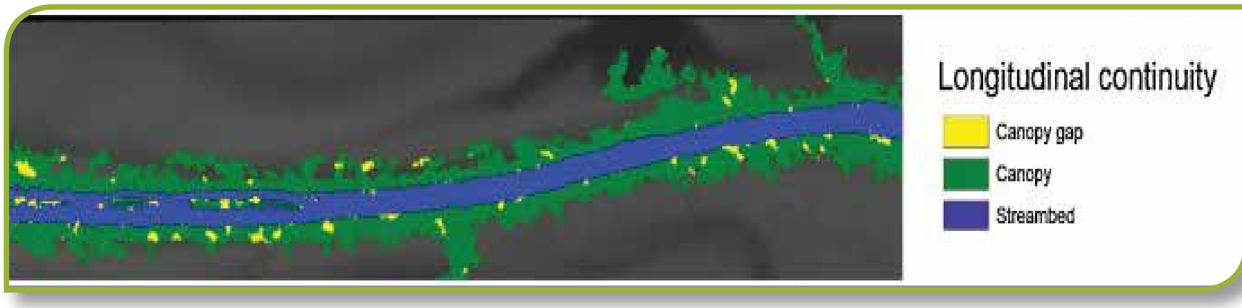

Figure 35: Longitudinal continuity can be estimated using LiDAR data by identifying zones in which there are canopy gaps (Johansen *et al.,* 2010).

Total width of the riparian buffer strip

In an agricultural context, it may be necessary to consider the total width occupied by vegetation along the stream, without restricting the analysis to the riparian zone. This indicator is important for assessing the capacity of a riparian zone to filter pollutants, thereby protecting water and aquatic habitat quality. This indicator should ideally be examined in relation to other variables, such as slope, soil type and type of agriculture adjacent to the riparian zone (Dosskey *et al.*, 2008; Tomer et al., 2009). For example, on a deeply incised stream, the riparian zone will be narrow, and the area occupied by vegetation between the cropland and stream (riparian buffer strip) will be wider.

Total plant biomass (aboveground)

Total plant biomass is often an indicator of system organization, resilience and vigour, and therefore, health. LiDAR can be used to obtain a relatively precise quantification of total plant biomass (R2=0.91) (Lefsky *et al.*, 2002).

Abundance of dead wood and woody debris

Dead wood and woody debris have a number of functions in terrestrial and aquatic environments. Recent studies have shown the potential of LiDAR to assess the relative abundance of dead wood in forests (Bater *et al.*, 2009). According to Hall (2009), it is also possible to use LiDAR to quantify the abundance of coarse woody debris in streams. The forms of dead wood and woody debris can easily be measured in the field (Quinn *et al.*, 1997a).

Vegetation overhang

Vegetation overhang corresponds to the horizontal projection of vegetation above the stream. LiDAR technology quantifies the amount of shade that is produced directly over the stream, an important structural attribute for avoiding excessive heating of the water during hours of high solar radiation (10:00 a.m. – 2:00 p.m.) (Johansen *et al.*, 2010) (Figure 37).

Canopy opening

Canopy opening is expressed as the percentage of sky that can be seen through the canopy. This indicator is often used in studies relating vegetation cover and water temperature or algal growth (Quinn *et al.,* 1997b). It can also be used as an indicator of the risk of invasion by shade-intolerant exotic plants (Parendes and Jones, 2000; Fortier *et al.,* 2011). Canopy is estimated using LiDAR or measured directly in the field using a hemispheric camera (Sasaki *et al.*, 2008).

Relative abundance of wetland trees

This measure provides information on the composition of the tree layer. The species are indicators of moisture and, as a result, of the extent of the riparian zone, as well as biophysical connectivity. The species identified in the field and their relative abundance can be related to the area occupied by the riparian zone. The capacity of LiDAR to provide the spectral signature of certain wetland tree species should be assessed.

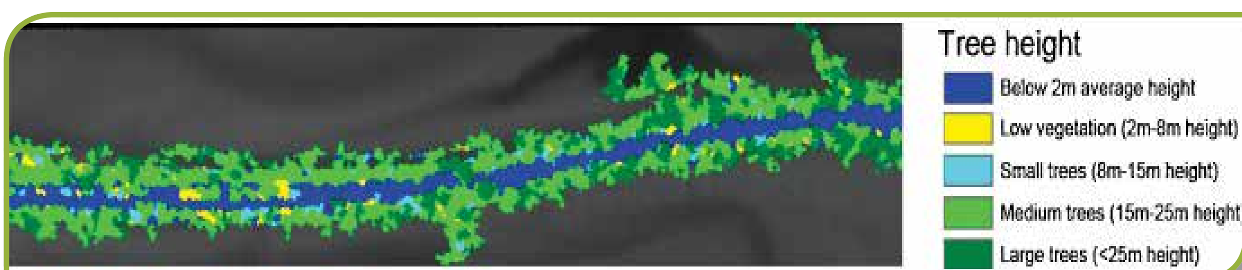

Figure 36: Example of a mapping using LiDAR data that identifies vegetation layers as a function of cover height (Johansen *et al.,* 2010).

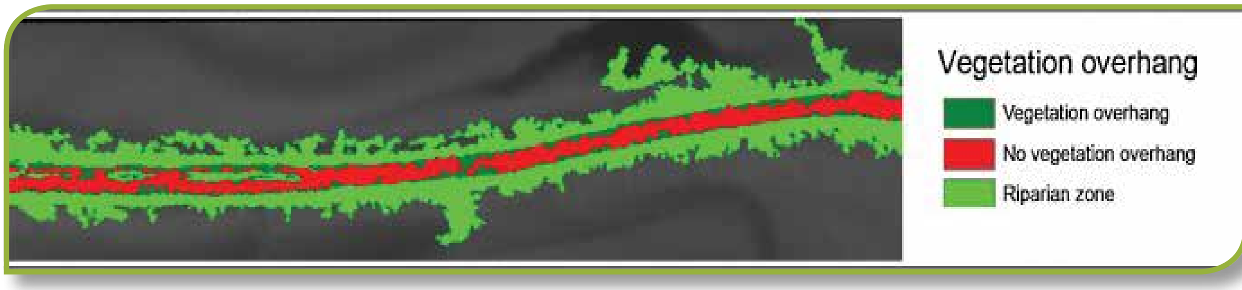

Figure 37: Vegetation overhang mapped using LiDAR technology (Johansen et al., 2010).

6.4.3. Indicators of stress

Portion of the riparian zone adjacent to annual crops
Given that annual crops (soybeans, corn) are a major stressor for the riparian zones, the percentage of the riparian zone located directly adjacent to annual cropland should be calculated for each reach of stream. It is important to assess the possibility of obtaining the spectral signature of the main annual crops.

Portion of the riparian zone adjacent to an unfenced pasture

In the absence of fences along streams, livestock is a significant source of stress for riparian zones. This indicator could therefore be calculated in the same way as the preceding indicator. The possibility of obtaining the spectral signature of pasture vegetation and a fence should be assessed.

Abundance of common reed *(Phragmites australis)*

Common reed is one of the key invasive plant species that invade riparian zones in Quebec agricultural landscapes. This species forms dense mono-species colonies that exclude native vegetation. The spectral signature of common reed is known, which means it is possible to map its distribution using LiDAR (Gilmore *et al*., 2008) (Figure 38).

Proportion of the riparian zone affected by flow control works

Various impacts are associated with flow control works and they are felt throughout the riparian system (Ward and Stanford, 1995) (Figure 27). In the Prairies, the construction of dams for irrigation purposes has led to the collapse of riparian forest communities downstream from the dams (Rood and Mahoney, 1990; Cordes *et al.,* 1997). Hydrographic maps can be used to identify streams whose flow is regulated by dams or other works.

Proportion of the riparian zone affected by morphological changes

Morphological changes to stream banks and channels can have major repercussions on water and terrestrial and aquatic habitat quality. Old aerial photographs and records regarding drainage work and the straightening of agricultural stream banks in Quebec could be used to quantify the scope of disturbances of the agricultural riparian zone (Beaulieu, 2001).

6.4.4. Indicator of aquatic habitat

Thermal regime of streams

According to the studies presented in section 5, water temperature is one of the only indicators of aquatic habitat that can be easily related to riparian vegetation structure. Various indicators of the thermal regime of streams could, however, be used (Chu *et al.*, 2010). This indicator must be measured directly in the field.

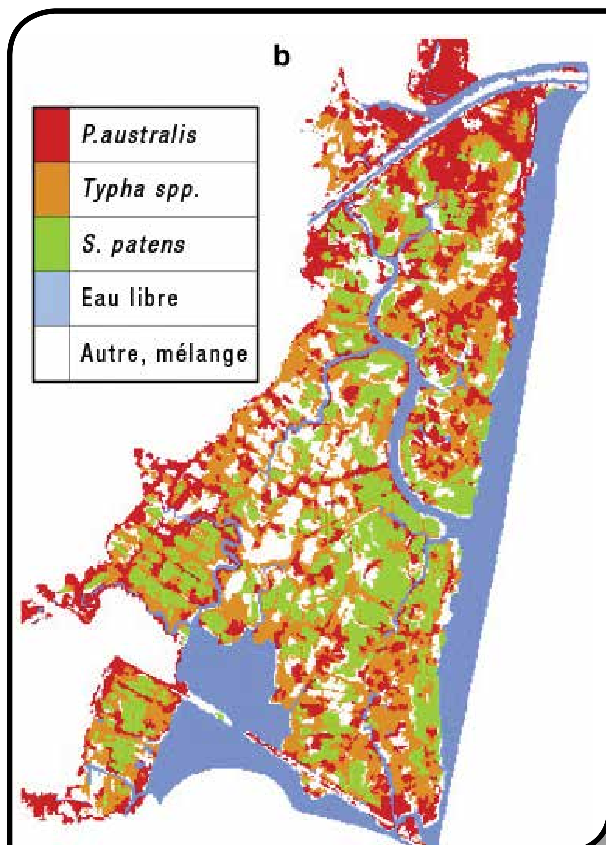

Figure 38: Map produced using LiDAR of various aquatic plant species (Gilmore *et al.*, 2008).

6.5. Spatialization of the indicator of agricultural riparian health

Given that riparian vegetation composition and structure vary depending on stream order (Gregory *et al.,* 1991), there may be some merit in assessing the agricultural riparian health of various stream reaches on the basis of Strahler stream order (Strahler, 1964). Moreover, since riparian vegetation differs from one ecoregion to another in Canada (Table 4.2), it is important to determine what health indicators are the most closely related to terrestrial and aquatic habitat quality in the various ecoregions. As mentioned earlier, the concept of agricultural riparian health is highly subjective in that it comprises an important social component. This aspect will have to be taken into consideration since the concept of agricultural riparian health likely differs from one region of Canada to another. In short, it appears to be important to perform a prior hierarchical and ecological classification of the various stream reaches that will be studied (Norris and Thoms, 1999).

In the case of indicators based on riparian conditions that are influenced by geomorphology and hydrology (Bendix and Hupp, 2000), a selection of streams that cover a range of geomorphologic and hydrologic conditions (Figure 13) will make it possible to test the robustness of the indicators with the different conditions found in Canada's agricultural riparian landscapes. In this sense, vertical stream erosion is a phenomenon that leads to a significant loss of agricultural riparian health, which requires the comparison of indicators on streams exhibiting various degrees of erosion.

Different agricultural practices generate different types of stress for the riparian zone. It may be possible to assess the health of the agricultural riparian zone in watersheds where the intensity of the agricultural practices differs (e.g., extensive vs intensive grazing, forage crops vs annual crops).

Lastly, decisions will have to be made about the type of riparian zones for which health assessments will be conducted (riparian zones of intermittent streams, drainage ditches and altered streams or natural permanent streams only).

6.6. Other comments and recommendations

A long list of agricultural riparian health indicators was created in section 6.4. However, it may well be that a number of these indicators are correlated with each other (e.g., vegetation cover height vs plant biomass; cover height vs vegetation overhang) and therefore that one is an indicator of the other. It is therefore suggested that a larger number of indicators be measured in order to then be able to conduct an analysis to identify informative, relatively uncorrelated indicators (that provide additional, new information). Certain indicators measured in the field could also be correlated to measurable indicators using LiDAR. For example, the extent of the riparian zone could be correlated to the abundance of wetland plants, which would make it possible to predict the abundance of wetland plants only with LiDAR data.

However, it is important to bear in mind that an agricultural riparian health indicator will not necessarily predict aquatic habitat quality, because the use of the area at the watershed scale is sometimes a better indicator of aquatic habitat quality than the local state of the riparian zone.

The assessment of the state of health of a riparian zone can be distorted when it is based on a comparison with the original natural state. The use of integral natural riparian zones as a reference system implies that it would be possible to restore all agricultural riparian zones to their original state. However, given that many agricultural riparian zones have deviated from their historical trajectory, it is not possible to compare them to integral natural systems. In other words, the remediation potential of agricultural riparian zones is not the same everywhere. Optimum operating points or desirable states of health must therefore be identified on the basis of the stresses and disturbances that altered the riparian systems. For that reason, the recommended assessment or management criterion is the level of ecosystem function (ecosystem functions or services normally provided by riparian zones), rather than ecological integrity (Jackson and Hobbs, 2009).

Another important word of caution concerns the use of a health index that encompasses various indicators (e.g., IQBR). As indicated by Suter (1993), this type of health index does not identify the components of the system that are degraded, and cannot be used to prescribe solutions to improve riparian health. However, it is possible to develop several health indicators that are not combined into a single index. Each indicator would assess one component of health, in order to provide a clearer picture of the true state of health of the agricultural riparian zone.

7. AGRICULTURAL RIPARIAN HEALTH AND OTHER AGRICULTURAL LANDSCAPE INDICATORS

7.1. Linkages between agricultural riparian health and agroforestry at the watershed scale

As stated in section 4.2.1, the presence of woody vegetation in riparian zones is an important sign of the health of the system due to the ecosystem functions and services it provides. As a result, agroforestry practices in riparian zones should have a positive effect on the state of agricultural riparian health. It is possible that increased agroforestry practices within a watershed may improve aquatic habitat quality, which is often correlated with total forest cover in the watershed.

7.2. Relationship between riparian health and wildlife habitat in agricultural landscapes

In view of the health components described in section 4, a healthy agricultural riparian zone would have the following characteristics: (1) a floodplain with a soil moisture gradient that varies according to flood conditions; (2) presence of significant tree cover over a large width of shoreline; (3) vegetation that interacts with the water table and stream during floods; (4) a herbaceous layer comprising few invasive alien plants and many aquatic herbaceous or facultative aquatic plant species; (5) a low level of human disturbance.

A riparian zone that has these characteristics will be able to support plant and animal species that are not found anywhere else in the agroecosystem (aquatic plants, waterbirds, amphibians, etc.) (Maisonneuve and Rioux, 2001; Boutin *et al.*, 2003; Deschênes *et al.*, 2003; Jobin *et al*., 2004). Healthy riparian zones can also contribute to protecting aquatic habitat by creating shade, providing organic debris, promoting channel development towards a more natural configuration (meanders) and trapping pollutants (sediment, nutrients and pesticides). Healthy agricultural riparian zones should increase the value of the wildlife habitat indicator at the landscape scale.

The presence of healthy riparian zones is important to terrestrial and aquatic animal communities because reciprocal flows of invertebrate prey link terrestrial and aquatic food webs (Baxter *et al.*, 2005) (Figure 39).

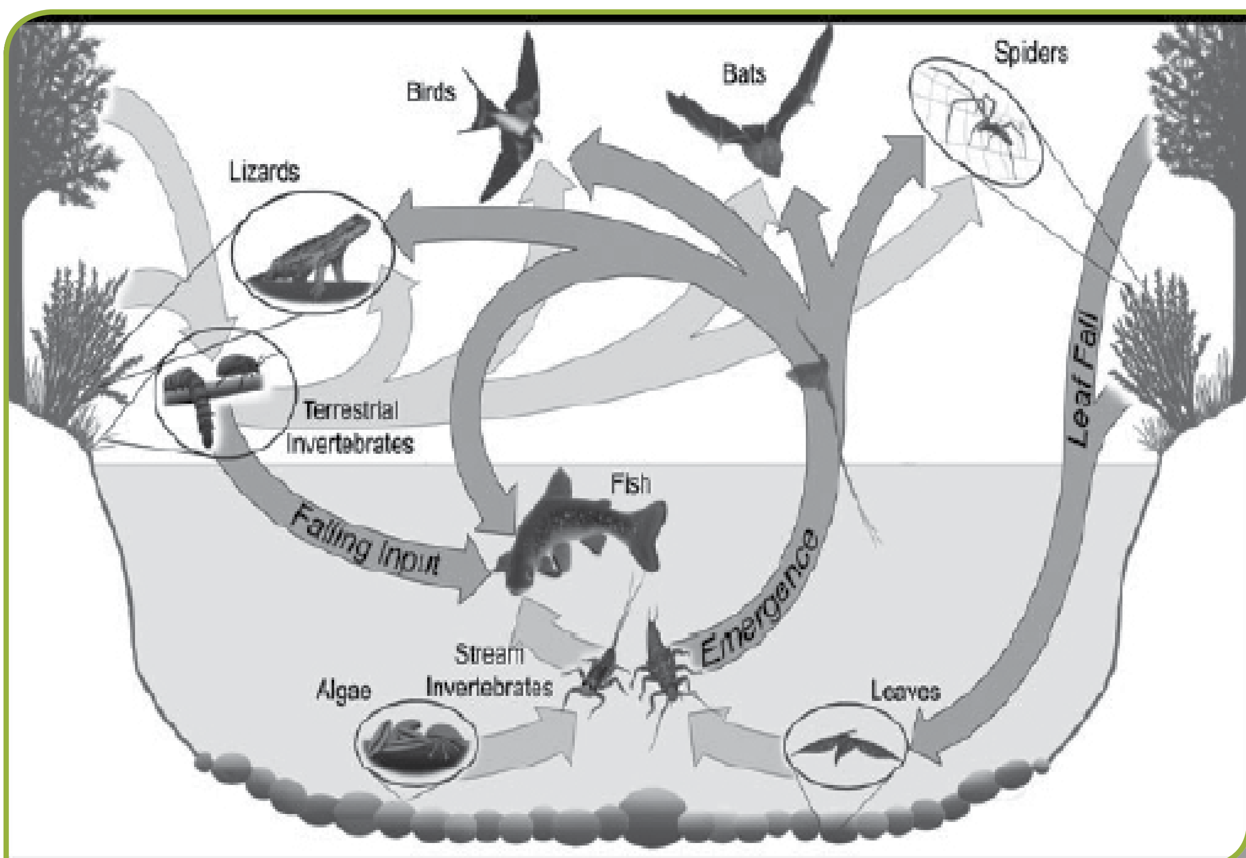

Figure 39: Conceptual diagram of flows of invertebrate prey and plant debris having a direct or indirect effect on aquatic and riparian food webs (Baxter *et al.*, 2005).

8. CONCLUSION

This report attempts to put the high level of complexity in the agricultural riparian zone into perspective. The development of an agricultural riparian health indicator should reflect this complexity or at least take it into consideration. This report also sheds light on certain challenges that may be encountered in developing an agricultural riparian health indicator, particularly if there is an effort to link this indicator to aquatic habitat quality. The components of riparian health were identified, as were several potential indicators.

This report does not cover the analysis of the cost-effectiveness ratio of implementing the identified indicators. The field methodology aimed at determining the reliability of the predictions of the state of riparian health using LiDAR technology is the subject of a separate report.

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APPENDIX 1

Trends observed in stressed ecosystems (from Odum, 1985).

System component affected	Response to stress or trends observed	Explanation or comment
Energetics	1. Community respiration increases.	Repairing damage caused by a disturbance increases community respiration.
	2. P/R (production / respiration) becomes unbalanced (< or > 1).	The R/R ratio tends towards 1 in undisturbed ecosystems. The ecosystem tends towards a state of balance.
	3. P/B and R/B (maintenance:biomass structure) ratios increase.	Energy is diverted from growth and production to repairing the damage caused by the disturbance. Maintenance of the system increases.
	4. Importance of auxiliary energy increases.	Productive energy is drained outside the system to dissipate entropy (disorder) and energy from the outside becomes more important in the continued survival of the system.
	5. Exported or unused primary production increases.	Unused resources are stored within the system or exported.
Nutrient cycling	6. Nutrient turnover increases.	Increased respiration increases mineralization of organic matter.
	7. Horizontal transport increases and vertical cycling of nutrients decreases.	The quantity of nutrients lost is greater than the quantity recycled.
	8. Nutrient loss increases.	Concrete examples in the agroecosystem: erosion of cultivated soils, fertilizer leaching.

System component affected	Response to stress or trends observed	Explanation or comment
Community structure	9. Proportion of «r» species increases. 10. Size of organisms decreases. 11. Lifespans of organisms or parts (leaves, flowers) decrease. 12. Food chains shorten. 13. Species diversity decreases and dominance increases in ecosystems with high diversity is high.	«R» species are fast growing, disperse themselve quickly and have a high reproduction rate. «R» species are opportunistic (ex : crop weeds). Tall organisms are replaced by smaller ones. E.g., pine forests exposed to air pollution lose their needles each year rather than every two years, and then the lifespan of the tree is reduced. Reduced energy flow at higher trophic levels. Small organisms out-compete large organisms. Predators are subject to bioaccumulation of toxins and are more sensitive to stress. In ecosystems with low species diversity, the reverse occurs.
General system-level trends	14. The ecosystem becomes more open. 15. Autogenic successional trends reverse. 16. Efficiency of resource use decreases. 17. Parasitism and other negative interactions increase, and mutualism and other positive interactions decrease. 18. Functional properties are more robust than are species composition and other structural properties.	Input and output environments become more important. Internal recycling is reduced. Internal processes that allow the system to evolve towards a state of greater complexity are disturbed. Late successional species are replaced by pioneer species. There is less complementarity between species. Parasitism increases. Cooperation and mutualism decrease due to the fact that there are more unused resources. Following stress, the system will evolve towards another state of homeostasis (internal equilibrium), but some species will disappear (e.g., disappearance of salmonids from a eutrophic lake).

APPENDIX 2

Ecosystem function and services associated with woody and herbaceous riparian vegetation (from: Wenger, 1999; Lyons et al., 2000; Paine and Ribic, 2002; Boutin et al., 2003; Jobin et al., 2004; Schultz et al., 2004; Altieri et al., 2005; Kelly et al., 2007; Mayer et al., 2007; Bentrup, 2008; Yuan et al., 2009; Dosskey et al., 2010; Fortier et al., 2010; Pärn et al., 2012).

Component affected	Function	Ecosystem service affected	Woody vegetation	Herbaceous vegetation
Water quality	Trapping of pollutants in the riparian zone.	Reduction of nonpoint source pollution and protection of aquatic habitat.	Woody and herbaceous vegetation are both effective at removing nutrients from groundwater and surface water, as well as trapping sediments in runoff. There are other factors that have a more significant influence on this aspect (e.g., topography, soil type, water table depth, aquifer characteristics).	
	Nutrient uptake and removal.	Reduction of nonpoint source pollution and protection of aquatic habitat.	Higher accumulation potential in the long term.	Very limited potential for nutrient accumulation if biomass is not harvested.
			Nutrient accumulation rate increases over the years.	Nutrient accumulation rate quickly levels off.
			Vegetation accumulates nitrogen and phosphorus, but is a temporary nutrient sink, because the sediment is made available by biological activity.	
	Transport and infiltration of water and pollutants into the root zone.	Reduction of flooding, nonpoint source pollution and erosion, protection of aquatic habitat, regulation of the water cycle.	Higher soil porosity beneath woody vegetation and therefore better water infiltration.	Lower soil porosity beneath herbaceous vegetation.
			Provides greater friction during floods.	Herbaceous vegetation is easily cut down to the ground.
			Higher evapotranspiration potential.	Lower potential evapotranspiration.
	Streambed stabilization.	Reduction of erosion, protection of aquatic habitat, reduction of nonpoint source pollution, regulation of the water cycle.	More effective at stabilizing deep banks.	More effective in stabilizing riparian surface soil.
			Produces wider, shallower streams.	Produces narrower, more deeply incised streams.

APPENDIX 2

Component affected	Function	Ecosystem service affected	Woody vegetation	Herbaceous vegetation
Water quality			Woody debris increases stream roughness, which reduces the erosive force of streamflow and promotes sediment removal in the stream.	Does not create accumulations of debris, which have an important impact on hydrology.
			Accumulations of woody debris promote the retention of sediment in streams. They create local flooding, resulting in sediment accumulation in the floodplain.	
	Transformation and removal of pollutants in streams.	Reduction of nonpoint source pollution and protection of aquatic habitat.	By promoting the accumulation of sediment, accumulations of woody debris promote denitrification.	Herbaceous vegetation enhances light availability in streams, thereby promoting the growth of algae and aquatic plants that remove nutrients.
			Production of large quantities of allochthonous organic matter promotes nutrient removal through the biofilm that colonizes the organic matter.	
			The creation of wider streams increases the reactive area within streams for transformation and removal of pollutants.	
	Physical barrier to pesti-cide drift.	Reduction of nonpoint source pollution, protection of aquatic habitat.	Because of its height, woody vegetation is effective in limiting aerial pesticide drift to streams.	Herbaceous vegetation is ineffective at reducing pesticide drift.
Soil quality	Source of organic mat-ter for soil.	Source of organic matter for soil.	Greater source of organic matter for soil due to greater biomass accumulation.	Less significant source of organic matter for soil.
			Higher potential for organic matter accumulation due to slower decomposition.	Lower potential for accumulation due to more rapid decomposition of organic matter derived from herbaceous species.

APPENDIX 2

<table>
<tr><th>Component affected</th><th>Function</th><th>Ecosystem service affected</th><th>Woody vegetation</th><th>Herbaceous vegetation</th></tr>
<tr><td rowspan="5">Terrestrial biodiversity</td><td rowspan="3">Creation of microclimates and habitats.</td><td>Refuge for terrestrial biodiversity.</td><td>Forest cover creates a microclimate favourable to forest and edge species.</td><td>Herbaceous cover creates a microclimate that favours pioneer and ‘open-habitat’ species, such as bobolink, which nest in grassy pastures.</td></tr>
<tr><td rowspan="2">Refuge for beneficial species that are crop pollinators and natural enemies of crop pests.</td><td colspan="2">Herbaceous vegetation and wooded areas on cultivated field edges provide overwintering and breeding sites, and a source of prey and nectar for beneficial arthropods in crops. Uncultivated land, such as forests, grasslands, old fields and herbaceous or woody hedges generally support more beneficial insects than cropland.</td></tr>
<tr><td>Forest cover limits invasion by invasive plants, because a large number of species are shade intolerant.</td><td>Non-forested banks are more often invaded by terrestrial or aquatic invasive plants.</td></tr>
<tr><td>Creation of corridors.</td><td>Refuge for terrestrial biodiversity.</td><td>Forest corridors can be used for movements of forest and edge species between forest fragments in agricultural landscapes.</td><td>Herbaceous corridors can be used for movements of open-habitat species.</td></tr>
<tr><td>Source of deadwood.</td><td>Refuge for terrestrial biodiversity.</td><td>Woody vegetation generates dead wood that is used by a number of terrestrial species.</td><td></td></tr>
<tr><td rowspan="4">Aquatic biodiversity</td><td rowspan="2">Creation of aquatic habitat.</td><td rowspan="2">Refuge for aquatic biodiversity, protection of species prized by recreational and commercial fishers.</td><td>Course woody debris creates fish habitat.</td><td></td></tr>
<tr><td>The creation of a wider stream channel by woody vegetation produces higher quality habitat.</td><td>The creation of a channel subject to vertical erosion by herbaceous vegetation is often associated with a loss of habitat quality.</td></tr>
<tr><td rowspan="2">Creation of a microclimate.</td><td rowspan="2">Protection of water quality, refuge for aquatic biodiversity, protection of species prized by recreational and commercial fishers.</td><td>Forest cover creates shade and prevents heating of water.</td><td>Herbaceous vegetation is responsible for greater variability in the thermal regime of streams and in an increase in average temperatures.</td></tr>
<tr><td>By trapping light, forest cover inhibits growth of algae.</td><td>Herbaceous vegetation promotes algal blooms in summer.</td></tr>
</table>

APPENDIX 2

Component affected	Function	Ecosystem service affected	Woody vegetation	Herbaceous vegetation
Aquatic biodiversity	Maintenance of trophic links.	Refuge for aquatic biodiversity and protection of species prized by recreational and commercial fishers.	Woody vegetation produces large quantities of allochtonous organic matter required for maintaining complex food webs.	Herbaceous vegetation is a poor source of allochtonous organic matter.
	Water treatment.	Refuge for aquatic biodiversity and protection of species prized by recreational and commercial fishers.	The various positive influences of the two types of vegetation on water quality will have beneficial effects on aquatic habitat quality.	
Adjacent agricultural land	Physical barrier to wind.	Reduction of wind erosion, soil formation, livestock protection, food production.	Trees generally have a greater windbreak effect.	Herbaceous plants have a limited effect as windbreaks.
	Creation of habitats.	Refuge for species that are crop pollinators and natural enemies of pests.	Woody vegetation provides more complex habitat that supports more crop pollinators and natural enemies of pests.	Herbaceous vegetation can serve as habitat for pollinators and natural enemies.
	Creation of shade.	Protection of livestock.	Trees produce a large amount of shade.	Herbaceous species create little shade.
Whole agricultural system (landscape)	Maintenance of the complexity at the landscape scale.		In an agricultural matrix, woody vegetation increases structural, functional and compositional complexity.	Natural herbaceous vegetation increases compositional diversity.

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