Environment Canada Environnement Canada

How Much Habitat is Enough?

Third Edition

About the Canadian Wildlife Service

Environment Canada's Canadian Wildlife Service is responsible for wildlife matters that are related to species of federal concern. This includes protection and management of migratory birds as well as nationally significant wildlife habitat, endangered species, control of international trade in endangered species and applied science on wildlife issues of national importance.

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http://www.ec.gc.ca/Publications/default.asp?lang=En&xml=1B5F659B-B931-4F37-A988-3DD73DF656B7

May be cited as:
Environment Canada. 2013. *How Much Habitat is Enough? Third Edition*. Environment Canada, Toronto, Ontario.

Print version
Cat. No.: CW66-164/2013E
ISBN 978-1-100-21922-6

PDF version
Cat. No.: CW66-164/2013E-PDF
ISBN 978-1-100-21921-9

Aussi disponible en français

Executive Summary

An important challenge for Canadian society in addressing the natural environment, one of the three elements of sustainability along with society and economy, is to ensure that there is adequate wetland, riparian, forest and grassland habitat to sustain minimum viable wildlife populations and help maintain selected ecosystem functions and attributes. The goal of the *How Much Habitat is Enough?* series is to provide science-based information and guidelines related to these natural systems and the biodiversity for which we all are the stewards.

The information is tailored to be of use to government and non-government restoration practitioners, planners and others involved in natural heritage planning, conservation and preservation.

The *guidelines* concentrated on providing information to assist decision makers in Great Lakes Areas of Concern in the setting and achievement of delisting criteria concerning fish and wildlife habitat beneficial-use impairments, and for post-delisting to provide further guidance on habitat restoration. An assessment of the *guidelines* (First Edition) in 2002 demonstrated that they were well-used within, but also outside of, Areas of Concern. The assessment indicated that they were indeed used as originally envisioned—as a guide to set restoration targets and restoration project locations, and also as a science-based reference for natural heritage practitioners.

A Second Edition was prepared and released in 2004. Regular updates are necessary to ensure that current research and scientific literature is incorporated and to provide an opportunity to expose new ways of considering old problems. This has also provided Environment Canada with an opportunity to invite contributions from a wider selection of topic experts.

In this, the Third Edition, there are 21 wetland, riparian, forest and grassland habitat guidelines and accompanying rationales. These are typically supported with a discussion addressing additional potential guidelines or issues. The grassland section is a new chapter that represents a first attempt at addressing these complex habitats. More than any other habitat type, grasslands are very dependent on human activity. This section is sure to evolve in the future as many of the newly listed species of conservation concern are found within grassland habitats. In the forest section, there has been an important shift toward a risk-based approach, one that is likely to be applied in other habitats in future years. Overall, the *guidelines* have expanded significantly both in terms of the supporting discussions within each topic and the base of scientific literature that has been reviewed.

The framework of guidelines and supporting text that comprise *How Much Habitat is Enough?* are effectively an "open file" meant to be built upon and to be adapted according to historical and current local conditions. This framework will hopefully continue to serve as a starting point to develop strategies to conserve habitat, develop natural heritage systems and discuss research needs around those habitats. This is in the spirit of keeping common species common, while restoring ecosystems for those species most at risk—the sensitive, the endangered, the threatened and the rare.

Acknowledgements for the Third Edition

How Much Habitat is Enough? (*A Framework for Guiding Habitat Rehabilitation in Great Lakes Areas of Concern*) (1998) was initially guided and championed by the Ontario Ministry of the Environment, the Canadian Wildlife Service of Environment Canada and the Ontario Ministry of Natural Resources. Subsequently, it has evolved into a living document in response to challenges, questions, experience and information brought forward by an active cross-section of individuals, agencies and organizations. These numerous contributions are greatly appreciated and acknowledged with this third edition.

This new edition further adds to the extensive list of individuals who have worked on this publication since its beginning in 1995 and on the first and second editions (1998, 2004). There have been many authors, contributors and reviewers over the past 18 years drawing from private consultants, federal and provincial agencies, Conservation Authorities, academia, and conservation organizations. Funding support has come from the Great Lakes Sustainability Fund of Environment Canada, the Ontario Ministry of the Environment and the Canadian Wildlife Service of Environment Canada as well as much support in the form of many in-kind reviews and advice.

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Table of Contents

1. How Much Habitat is Enough? Third Edition Introduction

How Much Habitat is Enough? provides science-based guidance to conserve and restore habitat for migratory birds, species at risk and other wildlife species within the settled landscapes of the lower Great Lakes and Mixedwood Plains. This biophysical area crosses provincial and international borders: in southern Ontario it forms an area south and east of the Canadian Shield.

The landscape of much of the world has irreversibly changed with increasing competition for land and resources. In many areas, the sustainable management of landscapes that considers the natural environment, society and economy is now commonly accepted as the goal in both land use planning and policy (North-South Environmental 2010). *How Much Habitat is Enough?* plays a role in that discussion by providing guidance for the restoration and conservation of wildlife habitat: in particular habitat of wildlife species under Environment Canada's mandate. In order to contribute best, general guidance is provided at the scale of habitat patches, watersheds and subwatersheds, reflecting the scale at which land use planning occurs in the settled landscapes of southern Ontario. The guidelines in this report are a non-exclusive contribution as to how that landscape could look, reflecting one approach at a particular scale.

The guidelines address the size and configuration of habitat patches and the overall quantity of habitat across a landscape for multiple species. It is an inclusive approach that allows for increased conservation of ecosystems. Such a systematic approach better captures the complexity of life and the multiple and often unknown linkages that allow species to flourish.

The Third Edition of *How Much Habitat is Enough?* contains new and updated science that has been used to provide new guidance or revise existing guidelines. This edition builds on previous versions, combining and removing some guidelines to reflect better the current body of knowledge. And where there is now a sufficient amount of supporting science, requests for guidance on topics such as grassland habitat have been addressed.

How to use this guide: Principles and considerations

How Much Habitat is Enough? was developed through the 1990s and published in 1998 as *A Framework for Guiding Habitat Rehabilitation in Great Lakes Areas of Concern*. This framework, comprised of guidelines and supporting information, has been used extensively for land use, habitat restoration and land securement planning as well as for policy development, as a post-secondary classroom resource and as a general conservation primer. The basic principles the framework was built upon have endured and lessons have been learned over the years. It is essential to consider these principles and lessons before applying the guidelines.

The basics

There are four habitat categories addressed by the guidelines: wetlands, riparian and watershed, forests, and grasslands. In reality these habitats overlap and are separated only to provide clear guiding principles. For example, a swamp could simultaneously provide forest- and wetland-related ecological functions and can be considered as forest and as wetland under these guidelines.

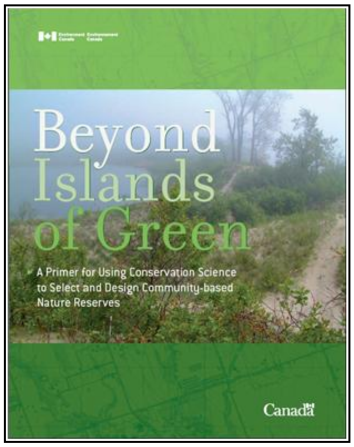

Figure 1. A conservation science primer from the Canadian Wildlife Service

Also, the individual guidelines and principles should not be applied separately but as a suite of interacting and interdependent guidance. For example, guidance on grassland patch size is best addressed in context of the overarching suggestion to target existing and potential grassland landscapes.

Where habitats are not defined, please refer to the community class descriptions found within Lee et al. (1998).

Science

The guidelines are based on scientific literature with an emphasis on published and unpublished studies from eastern North America, south of the Boreal Shield, and in particular the lower Great Lakes region. Supporting information and references are provided with each guideline, and the reader is urged to refer to the original studies and to adapt, and not necessarily adopt, the guidelines. Before applying the guidelines, consider that the advice provided emphasizes the habitat needs of migratory birds and other species of federal concern, such as species at risk, and that the guidelines also reflect the limits of existing scientific knowledge. To help bridge knowledge gaps regarding species of federal concern, science on species that are not specifically under the federal mandate has also been used to help formulate the guidelines.

Regardless, restoring and conserving habitat using the guidelines will contribute to the maintenance of multiple ecological functions and health across various mandates, along with associated derived goods and services for humans. This is discussed further in the text box "Why birds?" Additional general guidance regarding conservation biology and landscape ecology is provided by sources such as Environment Canada's *Beyond Islands of Green* report (www.ec.gc.ca/Publications/default.asp?lang=En& xml=1 CEC82DE-8E6C-4A51-A0A5-48A2A6FD078D), and information on the needs of migratory birds is provided in Bird Conservation Region (BCR) Strategies (www.ec.gc.ca/mbc-com/default.asp?lang=En&n=1D15657A-1).

Guidelines are not binding targets

The framework is not legislative and should be viewed as a means to guide, not dictate, local decisions, by providing planners, rehabilitation teams and other decision makers with the best available information. Direction regarding habitat for specific species at specific locations is provided through various provincial and federal policies and statutes including the federal *Species at Risk Act* and *Migratory Birds Convention Act, 1994* and Ontario's *Endangered Species Act, 2007, Fish and Wildlife Conservation Act* and *Planning Act*. The general guidance provided within *How Much Habitat is Enough?* is secondary to direction provided under such statutes.

Conserve it first

The conservation of existing habitat must remain the most important ecological planning activity in any jurisdiction. Restoration will always be necessary to have a fully functioning natural heritage system in degraded landscapes, but protecting existing habitat is far more efficient and more effective.

The guidelines are minimums

The *How Much Habitat is Enough?* guidelines are intended as *minimum* ecological requirements with the objective to maintain wildlife populations at levels that would prevent local extirpations of species. However, the preference is to maintain population levels to provide better for long-term species persistence: this preference is reflected in the Third Edition in the "percent forest cover" guideline, which has been rewritten to address risk associated with different forest cover amounts. Generally, a greater diversity and amount of habitat than the minimum will almost always be more beneficial in terms of supporting healthy species populations and a wider range of ecological functions. It should be noted that *How Much Habitat is Enough?* should not be used to set lower habitat targets for landscapes that

Why birds? An example of the limits of scientific knowledge and the value of focal species

Forest birds are often used as indicators of the quality of the landscape because they are more easily surveyed, and more is known about their habitat requirements and distribution than other wildlife groups. Likewise, the protection and regulation of migratory birds is a major part of the Canadian Wildlife Service's mandate. The guidelines in *How Much Habitat is Enough?* reflect the general minimum habitat requirements of the different bird pillars or guilds, encompassing priority species within the Ontario portion of BCR 13. Where there are specialized habitat requirements for individual species, these needs are noted in the *Bird Conservation Strategy for Ontario's Bird Conservation Region 13: Lower Great Lakes/St. Lawrence Plain* (Environment Canada 2012) or within relevant recovery strategies or action plans (see the SARA registry: www.sararegistry.gc.ca).

Less is known about the sensitivity of invertebrates, amphibians, reptiles, plants and small mammals to, for example, forest fragmentation, although amphibians and reptiles have been receiving more research attention over the past decade or so. In the absence of this knowledge, forest birds serve as a useful indicator of forest and ecosystem health. They can be seen as "umbrella species," a term debated within the field of conservation science that refers to species with requirements that overlap requirements for many other species.

currently have habitat in excess of the guideline minimums, as often irreversible ecological damage will occur. Removing or degrading habitat, for example moving from 50 to 30% forest cover, will result in reduced wildlife populations, reduced capacity to provide ecological goods and services, and generally lower ecological integrity. The guidelines should be a starting point: where local decision-making can provide more habitat than the minimum, then a greater robustness in natural heritage systems can be anticipated.

Adapt first, adopt second

The guidelines are general and not landscape- or watershed-specific. They should be viewed as ecological principles that detailed local scientific and other knowledge, including traditional knowledge, will build upon and inform where appropriate. Local ecological differences such as the location of historic grasslands or original extent of wetlands, and whether precedence will be given to forest, wetland or grassland habitats, need to be considered where appropriate.

Look beyond local boundaries

Habitat within a planning unit, such as a watershed or municipality, is often ecologically connected with features beyond that unit. In order to promote linkages of habitat between watersheds and across landscapes, surviving habitat corridors and geographic features should be carefully considered. Likewise, local efforts should consider larger-scale planning efforts, such as conservation blueprints and conservation action plans, unique biogeographic features such as the Oak Ridges Moraine, and the role the local landscape plays in the overall diversity and integrity of larger ecoregions and ecozones, such as the Mixedwood Plains. Projects such as the Carolinian Canada Coalition's "Big Picture" illustrate the need to look at larger ecosystems and landscapes.

Consider landscape context

If a planning unit spans across different ecoregions, then consider representing the full range of natural communities that occur in those ecoregions. For example, concentrating habitat conservation and restoration in only one part of a large watershed may result in an under-representation of communities and species in other ecoregions within that watershed.

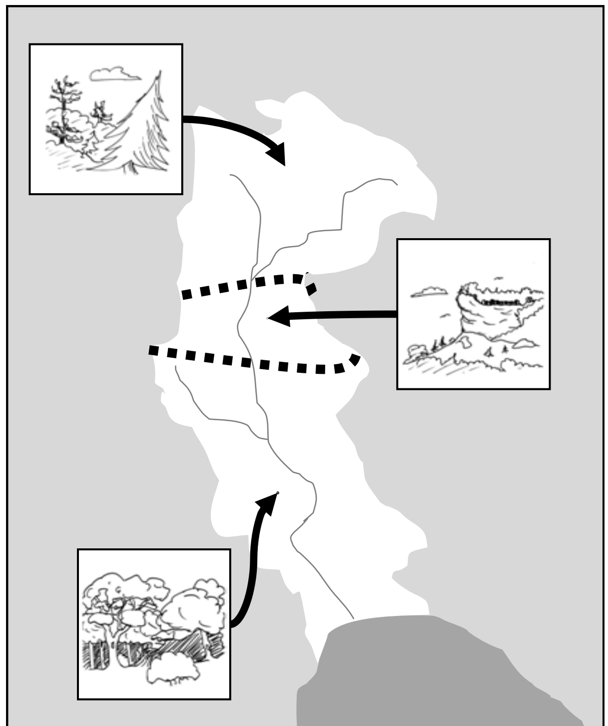

Figure 2. Multiple ecoregions in one watershed jurisdiction

Acknowledge stressors beyond habitat

There are additional stressors that affect fish and wildlife populations beyond the loss of habitat, including toxins, nutrient enrichment, disease and invasive species. In several cases, species are at risk not because of direct habitat loss but because of other stressors such as invasive pathogens or species (e.g., Butternut, American Chestnut and ash trees).

The importance of the matrix

The "matrix" in landscape ecology typically refers to land cover or land covers that dominate a landscape. In the Mixedwood Plains, the rural areas outside of large cities and towns are predominantly agricultural, dominated by open farmland and interspersed with natural features and small towns and villages. The farmland may be active, lying fallow or regenerating to forest, wetland or other "natural" land covers. In large cities and towns, a built-up urban matrix dominates. In the Mixedwood Plains and other settled landscapes, protected natural areas, natural heritage features (such as forest patches) or natural heritage systems (interconnected or linked natural features) are embedded within these urban and agricultural matrices.

Earlier scientific literature largely disregarded the influence of the matrix on species within natural areas in fragmented landscapes. However, there is a mounting body of evidence suggesting the nature of the matrix can have a profound effect on habitat use by different species, particularly in highly fragmented landscapes. It can do this by (a) directly influencing the dynamics within the natural habitat patch itself, and (b) influencing the ability of species to move between patches (Ewers and Didham 2006). Studies to date in fragmented landscapes have shown that effects are generally greater in urban than in rural areas (Borgmann and Rodewald 2004; Dunford and Freemark 2004; Hansen et al. 2005), and that successional habitats in rural landscapes can provide important supporting habitats for forest species (Gibbs et al. 2005; Milne and Bennett 2007). This means that in much of the Mixedwoods Plains Ecozone, the attributes of the matrix can be as important in influencing species composition and abundance as the attributes of the natural habitat patches themselves.

The Third Edition: Emerging considerations

Some of the following emerging concepts were noted during the preparation of the Third Edition.

Acknowledge the limits of land use planning, restoration and protection

The guidance in this report is aimed at a common framework of conservation measures that include land use planning, habitat restoration and protecting natural areas. However, there are forces, such as commodity markets, that influence the landscape and do not easily fit within this conservation framework. For example, agricultural practices create habitat for grassland birds in the form of hay or pasture. These practices are driven by factors such as market forces and consumer demand. Land use planning can allow uses such as planting hay and grazing, but generally it cannot force land to be used in a particular way.

Species at risk

The guidelines are intended to help keep common species common, and the general guidance provided should not be seen as a substitute for identifying specific species-at-risk habitat under the *Species at Risk Act* (Government of Canada 2002) or regulated habitat under the *Endangered Species Act, 2007* (Government of Ontario 2007). It is intended that the guidance in this report will help to protect and conserve species at risk by restoring ecosystems upon which those species rely.

Reference points, populations and change

The current settled landscapes of northeastern North America have undergone great change since before European contact. After the retreat of the great ice sheets and prior to European contact, southern Ontario was a forest biome with vast swamps and abundant marshes that supported extensive stream systems and abundant populations of forest and wetland species. There was an unknown extent of grassland habitats, many of which were created by the activities of First Nations. Subsequent to settlement by Europeans and with land clearance in the 19th century, an open country matrix arose. Today, the landscape continues to change along with species populations. *How Much Habitat is Enough?* looks to the dominant habitats associated with these time periods as multiple reference points for restoration efforts. The goal is to incorporate habitat requirements for a diverse suite of species into the current landscape, while factors such as climate change and invasive species continue to challenge and change ecosystems. The guidelines work on the principle that efforts to conserve and restore the extent and integrity of natural and surrogate habitats will increase ecosystem resilience and better allow species to adapt to evolving conditions.

A note about scales and planning units

The guidelines operate at different scales and are applied to different features to reflect specific ecological functions. The wetland and riparian guidelines tend to address the health of aquatic habitat within a watershed; therefore, watersheds are the unit and scale used within these guidelines. The forest and grassland guidelines tend to address terrestrial species needs. These needs are independent of watershed boundaries and are instead described in units such as patches and landscapes, as those are the units where ecological effects will be realized.

In addition, many of the guidelines are often described in terms of a watershed or municipal scale, regardless of ecological function, because watersheds and municipalities are common conservation planning units. An example of using units independently of function can be seen in one of the forest guidelines. The guideline calling for 30 to 50% forest cover applies to an area under 500 to 1000 square kilometres. This happens to be roughly the size of two major land use planning units: most rural Ontario municipalities and smaller fourth-order (quaternary) watersheds. While the guideline refers to terrestrial habitat independent of watershed function or municipal boundaries, a watershed or municipality is a similar-sized unit that is useful for planning purposes.

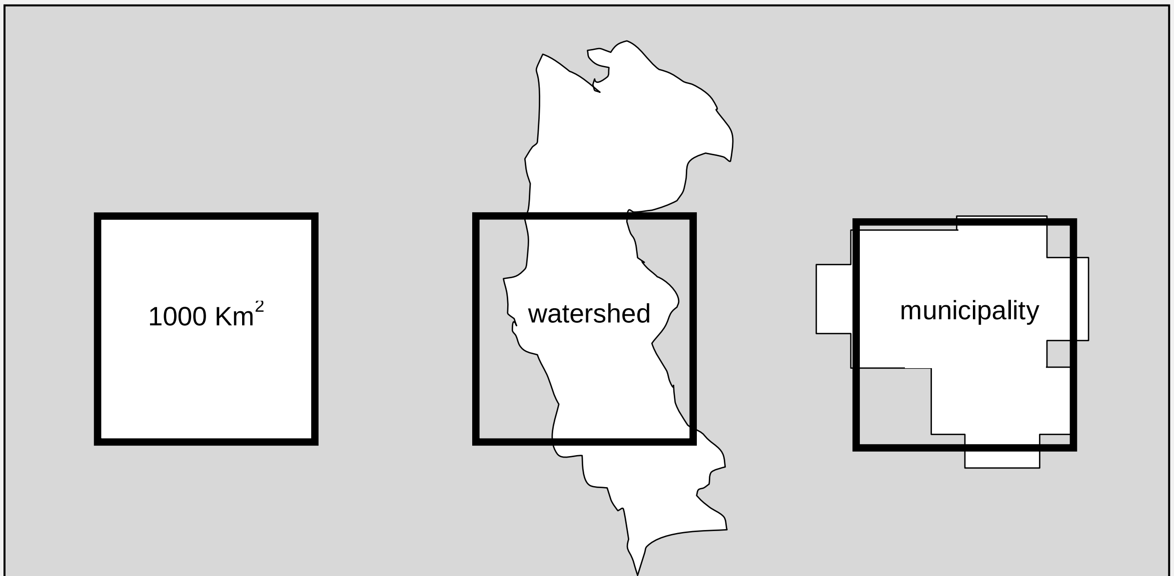

Figure 3. Similar-sized land units that can be used for planning purposes

There is also a hierarchy to some of the guidelines. In terms of the forest guidelines, when regional forest cover is below 30 to 50%, then guidelines addressing patch size and configuration gain importance. When forest cover is above 30 to 50%, then the configuration of the habitat is less important for maintaining species richness.

Urban areas

How Much Habitat is Enough? assumes a predominantly non-urban context. In some of these watersheds, changes to ecosystems have not been so drastic as to preclude rehabilitation of these systems. However, in many urbanized areas, ecosystems have shifted to an entirely new, and often irreversible, ecological state. It will therefore be impossible to fully implement these guidelines in urban areas. For example, a built-up environment will generally not be suitable for many area-sensitive forest birds. The most effective conservation approach would be to identify and protect habitat above minimum levels before urbanization occurs. However, this does not preclude increasing habitat cover in urban areas. Wetlands, forests, grasslands and riparian zones provide many vital ecological services both for wildlife and humans. Even partial implementation of the guidelines will yield ecological benefits. It may be far more appropriate to consider new baselines and targets for habitat in urban areas. There may also be discussions as to the need to compensate elsewhere in a region for habitat loss due to urbanization within that region, which would affect habitat targets set outside urban areas.


GREAT LAKES FACT SHEET

Forest Birds in Urban Areas: Habitat Needs of Area-Sensitive Species

Canada

Figure 4. A guide addressing habitat needs in urban areas

Climate change

Although there are many models and predictions, current data suggest that over the next 40 or so years there will be an average mean temperature increase of 2.5 to 3°C as well as an increase in annual levels of precipitation in the order of 10% (Expert Panel on Climate Change for Ontario 2009).

Habitats in North America and the species they support are already beginning to show responses to climate change. Documented responses include range shifts in some species, earlier calling for some amphibians, and longer stays on their breeding grounds by some wetland birds (2degreesC 2007; Crick 2004; Niven et al. 2009; Varrin et al. 2009).

However, for some of the Mixedwood Plains habitats the net impact upon extent and composition remains unclear. For example, some wetland species that are sensitive to small changes in hydrologic regimes may be affected. It is possible that some wetland species will expand their range or populations due to a generally warmer and wetter climate. The full effects of climate change remain to be determined, and as the climate change models become more refined it will be possible to better predict anticipated effects on a regional scale, and for different habitats.

A precautionary approach is to strive toward protecting and restoring more complete ecosystems with greater integrity that will be more resilient to change. This requires venturing beyond the minimum amounts of forests, wetlands, grasslands and riparian areas required to maintain species populations above extinction thresholds.

What about the Canadian Shield?

The environment of the Canadian Shield contrasts greatly with the settled landscapes of the Mixedwood Plains of southern and eastern Ontario. The Shield's complex terrain is characterized by rocky outcrops, relatively thin soils covering Precambrian bedrock, and an abundance of lakes and wetlands. It remains largely forested.

In this rugged landscape, with relatively low agricultural potential, the human population density never reached that of areas further to the south. Today, the Shield is in many regards an opposite of the Mixedwood Plains—cleared land and human settlements form patches within the forested matrix. In such an environment the approach taken and presented in *How Much Habitat is Enough?*—based as it is on science from the settled agricultural landscapes—does not transfer well to the Shield.

An area of particular interest to planners and ecologists alike is the southern part of the Shield. South of the French River and Algonquin Park is a distinct transition area bordering the Mixedwood Plains and the Great Lakes. This is an area of growing human influence with many growing urban areas and second homes and other residential and related development adding to existing land uses and activities such as forestry, mining and recreation.

In this region, a reasonable approach might be to consider how much the landscape can be disturbed before there are substantive ecological effects. This contrasts with the "*How Much*" approach used in settled landscapes that considers the minimum natural land cover that may be required to conserve biodiversity. This different approach was outlined in the unpublished report *How Much Disturbance is too Much?: Habitat Conservation Guidance for the Southern Canadian Shield* available from Environment Canada (Beacon Environmental 2012).

The report does not provide specific guidelines; rather, it assesses the available science and provides an initial review of the topic along with the introduction of general principles and concepts. As a foundation, *How Much Disturbance is too Much?* stresses the importance of identifying local and regional "habitat mosaics." In these defined areas, restricting most types of development should be considered. These areas would consist largely of forested areas with components of diverse wetlands (e.g., swamps and marshes, as well as fens) interspersed with open, shrub and treed rock barrens. These areas could be based on landscapes having relatively high levels of the following characteristics: diversity, naturalness and habitat extent. They could be identified at two levels: (1) Regional Habitat Mosaics and (2) Local Habitat Mosaics. Regional Habitat Mosaics could include blocks of habitat identified as important on a regional scale (e.g., within an ecodistrict or ecoregion) and would likely include many of the Crown Lands in the southern Canadian Shield. Local Habitat Mosaics could include blocks of habitat identified as important on a local scale (e.g., within a county or township) but similarly capture concentrations of diverse natural areas that are largely undisturbed. Local mosaics would contain one or more large blocks of habitat. Planning could be coordinated at both regional and local jurisdictional levels to identify opportunities for ensuring that Regional and Local Habitat Mosaics are complementary or in proximity to each other where possible, and where it makes sense within the given biophysical and land use context. These areas should cover 50 to 60% of a jurisdiction and be connected to one another. Disturbances within the mosaics should be avoided, especially disturbances created by roads of any kind.

Table 1. Summary of Wetland, Riparian, Forest and Grassland Habitat Guidelines

Parameter	Guideline
Wetland Habitat	
Percent wetlands in the watershed and subwatersheds	Ensure no net loss of wetland area, and focus on maintaining and restoring wetland functions at a watershed and subwatershed scale based on historic reference conditions. At a minimum, the greater of (a) 10% of each major watershed and 6% of each subwatershed, or (b) 40% of the historic watershed wetland coverage, should be protected and restored.
Wetland location in the watershed	Wetlands can provide benefits anywhere in a watershed, but particular wetland functions can be achieved by rehabilitating wetlands in key locations, such as headwater areas (for groundwater discharge and recharge), floodplains and coastal wetlands. Consideration should also be given to protecting networks of isolated wetlands in both urban and rural settings.
Amount of natural vegetation adjacent to the wetland	Critical Function Zones should be established around wetlands based on knowledge of species present and their use of habitat types. Protection Zones should protect the wetland attributes from stressors. Recommended widths should consider sensitivities of the wetland and the species that depend upon it, as well as local environmental conditions (e.g., slopes, soils and drainage), vegetative structure of the Protection Zone, and nature of the changes in adjacent land uses. Stressors need to be identified and mitigated through Protection Zone design.
Wetland proximity	Wetlands that are in close proximity to each other, based on their functions, or that are in close proximity to other natural features, should be given a high priority in terms of landscape planning.
Wetland area, shape and diversity	Capture the full range of wetland types, areas and hydroperiods that occurred historically within the watershed. Swamps and marshes of sufficient size to support habitat heterogeneity are particularly important, as are extensive swamps with minimum edge and maximum interior habitat to support area-sensitive species.
Wetland restoration	Focus on restoring marshes and swamps. Restore fens under certain conditions. For effective restoration, consider local site conditions, have local sources to propagate new vegetation, and wherever possible refer to historic wetland locations or conditions. Prioritize headwater areas, floodplains and coastal wetlands as restoration locations.
Riparian Habitat	
Width of natural vegetation adjacent to stream	Both sides of streams should have a minimum 30-metre-wide naturally vegetated riparian area to provide and protect aquatic habitat. The provision of highly functional wildlife habitat may require total vegetated riparian widths greater than 30 metres.
Percent of stream length naturally vegetated	75% of stream length should be naturally vegetated.

Parameter	Guideline
Percent of an urbanizing watershed that is impervious	Urbanizing watersheds should maintain less than 10% impervious land cover in order to preserve the abundance and biodiversity of aquatic species. Significant impairment in stream water quality and quantity is highly likely above 10% impervious land cover and can often begin before this threshold is reached. In urban systems that are already degraded, a second threshold is likely reached at the 25 to 30% level.
Forest Habitat	
Percent forest cover	30% forest cover at the watershed scale is the minimum forest cover threshold. This equates to a high-risk approach that may only support less than one half of the potential species richness, and marginally healthy aquatic systems; 40% forest cover at the watershed scale equates to a medium-risk approach that is is likely to support more than one half of the potential species richness, and moderately healthy aquatic systems; 50% forest cover or more at the watershed scale equates to a low-risk approach that is likely to support most of the potential species, and healthy aquatic systems.
Area of largest forest patch	A watershed or other land unit should have at least one, and preferably several, 200-hectare forest patches (measured as forest area that is more than 100 metres from an edge).
Forest shape	To be of maximum use to species such as forest breeding birds that are intolerant of edge habitat, forest patches should be circular or square in shape.
Percent of watershed that is forest cover 100 m from forest edge	The proportion of the watershed that is forest cover and 100 metres or further from the forest edge should be greater than 10%.
Proximity to other forested patches	To be of maximum use to species such as forest birds and other wildlife that require large areas of forest habitat, forest patches should be within two kilometres of one another or other supporting habitat features. "Big Woods" areas, representing concentrations of smaller forest patches as well as larger forest patches, should be a cornerstone of protection and enhancement within each watershed or land unit.
Fragmented landscapes and the role of corridors	Connectivity width will vary depending on the objectives of the project and the attributes of the forest nodes that will be connected. Corridors designed to facilitate species movement should be a minimum of 50 to 100 metres in width. Corridors designed to accommodate breeding habitat for specialist species need to meet the habitat requirements of those target species and account for the effects of the intervening lands (the matrix).
Forest quality – species composition and age structure	Watershed forest cover should be representative of the full diversity of naturally occurring forest communities found within the ecoregion. This should include components of mature and old growth forest.

Parameter	Guideline
Grassland Habitats	
Where to protect and restore	Focus on restoring and creating grassland habitat in existing and potential grassland landscapes.
Habitat type and area	Maintain, restore and create native grassland patches to their historic extent and type at a county, municipal and/or watershed scale considering past presence and current conditions.
Landscape configuration, heterogeneity and connectivity	Grassland habitat patches should be clustered or aggregated, and any intervening land cover should be open or semi-open in order to be permeable to species movement.
Patch size	Maintain and create small and large grassland patches in existing and potential local grassland landscapes, with an average grassland patch area of greater than or equal to 50 hectares and at least one 100-hectare patch.
Landscape heterogeneity	Some grassland habitat should be located adjacent to hedgerows, riparian and wetland habitats for species that require different habitat types in close proximity.

2. Habitat Guidelines

2.1 Wetland Habitat Guidelines

Wetlands are defined here following the Ontario Wetland Evaluation System of the Ontario Ministry of Natural Resources. Essentially, these are areas where hydrophytic (water-adapted) plants comprise 50% or more, by cover, of the vegetation in a given area, and when standing water is present, it is less than 2 metres deep.

Wetlands can provide valuable ecological and hydrological functions at both site-specific and watershed scales. Many of southern Ontario's flora and fauna inhabit wetlands during part or all of their life cycle, including many species at risk. Wetlands are known to be biologically diverse habitats, tending to support a wider range of flora and fauna than either temperate upland forests or grasslands, particularly on a species per area basis (Comer et al. 2005; Gibbons et al. 2006; Meyer et al. 2003).

Wetland types

Understanding the diversity of different wetlands is important for conservation planning and restoration because of their differing hydrologic regimes and vegetative structures, each supporting unique assemblages of species and combinations of ecological functions.

The wetland classification system for southern Ontario, which is the standard that has been in place since the 1980s, divides wetlands in the Mixedwood Plains into four types: bogs, fens, swamps and marshes.

Bogs are peat-accumulating wetlands that trap precipitation as the major water source. They typically have acidic organic soils, and often contain *Sphagnum* mosses and ericaceous shrubs (Ericaceae, a family of plants commonly found in acidic and infertile conditions).

Fens are peat-accumulating wetlands with groundwater as the dominant water source, which support a variety of plant species, including orchids, sedges and grasses.

Swamps are wetlands dominated by trees and shrubs, with periodic standing water, limited drainage, and often neutral or slightly acidic organic soils.

Marshes are wetlands that are almost always flooded and are characterized by a mixture of emergent, floating and submerged aquatic vegetation such as reeds, sedges, pondweeds and water lilies.

Figure 5. Derryville Bog, Durham Region, Ontario: one of very few true bogs in southern Ontario
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Bogs and fens

Bogs are highly specialized environments, and true bogs are rare in the southern part of the

Great Lakes basin. They receive almost all of their water and nutrients from precipitation, are acidic, have very low productivity and are dominated by plants that are adapted to low nutrient levels. Fens receive most of their water and nutrients from groundwater, and they may be either nutrient-rich or nutrient-poor, with nutrient-rich fens having a greater diversity of species while the nutrient-poor share many characteristics with bogs.

While they play major roles in the hydrology and ecology of the Boreal Shield Hudson Plains Ecozone, bogs and nutrient-poor fens are relatively rare and unique in the south and play less of a role in providing wildlife habitat than the larger, more extensive, and more diverse swamps and marshes of the Mixedwood Plains.

Swamps

Swamps have more than 25% tree or shrub cover; swamps are a major natural landscape feature because of their sheer extent (they comprise almost 90% of the remaining wetland area in southern Ontario). Swamps make significant contributions to forest cover and to watershed hydrology and aquatic health. They also disproportionately contribute to biodiversity because of the variety of cover types they provide. This includes providing mammal wintering areas, sources for a high proportion of cold-water streams, habitat for forest-interior or area-sensitive species, and habitat for many wildlife species, including numerous species at risk, often all simultaneously.

The diversity of swamps goes beyond their extent and cover types, and includes a temporal element that is key to the life cycle of many wildlife species. The plant communities of many swamps are dynamic and provide a variety of seasonal habitat attributes, with the understorey being dominated by wetland species early in the growing season, and plants adapted to drier conditions playing a greater role later in the year. In addition, spring flooding creates ephemeral ponds that are used for breeding by frogs, toads and salamanders. These same pools are also important breeding areas for invertebrates such as some caddisflies and midges, and these, in turn, are important food for bats and many bird species.

Figure 6. Flooded swamp at Minesing wetland
© *Graham Bryan*

Hydrologically, swamps shape the characteristics and health of lower-order streams; they moderate stream hydrology and help maintain water quality. This affects wildlife habitat all the way downstream to the receiving waters of species-rich coastal wetlands.

Marshes

Marshes are typically associated with the word “wetland”; however, they represent only about 10% of the area of wetlands in southern Ontario, and about 5% of all of the province’s wetlands (Riley 1988). Today, extensive marshes are rare in the landscape relative to historic conditions, so species that require this habitat are also restricted in their distribution. Several fish and wildlife species are totally dependent on marshes, and a high proportion of these are of provincial and federal concern. Some examples of obligate marsh species include Spotted Gar, Spotted Sucker, Banded Killifish, Bullfrog, Pied-billed Grebe, Red-necked Grebe, Least Bittern, Ruddy Duck, King Rail, Virginia Rail, Sora, Common Gallinule, American Coot, Forster's Tern, Black Tern, Marsh Wren and Muskrat.

The noteworthy ecological role of marshes is illustrated by Jude and Pappas (1992), who found that of 113 fish species occurring in the Great Lakes, 41.6% were coastal marsh species and 31% used coastal marshes for nursery habitat or feeding. In Lake Ontario marshes, 63.9% of species present used marshes for spawning and 86% of species used marshes as nursery habitat. The importance of marshes to the fish of the Great Lakes and also inland water bodies cannot be overemphasized. Approximately 90% of the fish biomass in Lake Erie is forage fish (also called prey or bait fish), and most of this is produced in wetlands (Keast et al. 1978; Stephenson 1990). Likewise, marshes are key to waterfowl nesting and stopover.

As with swamps, inland marshes help shape stream hydrology and aquatic ecosystem health, including playing a major role in improving and maintaining water quality, resulting in the maintenance of downstream wildlife habitat. Marsh vegetation also stabilizes shorelines and reduces the risk of erosion, protecting and stabilizing in-situ and adjacent habitat, along with reducing sediment delivery to water bodies (Sheldon et al. 2005). Marshes are also the primary building block for wetland restoration; establishing a marsh is often the initial step in the long-term process of establishing swamps and more complex and diverse wetland communities.

The following series of wetland habitat guidelines relate to the amount of wetlands and wetland location in a watershed, the amount of vegetation adjacent to a wetland, representation of wetland area and type, and wetland restoration, as summarized in Table 2.

Figure 7. Long Point – a Great Lakes Coastal Marsh
© *Environment Canada*

Wet meadow marshes – a specialized wetland type

Wet meadows provide habitat for a high diversity of plants and wildlife in southern Ontario. Rare species known to inhabit wet meadows include Henslow's Sparrow and Yellow Rail. Wet meadows are habitats subject to temporary flooding dominated by herbaceous plants typical of moist soils, and can be easily overlooked as upland habitats if seen during a dry period. These types of wetlands occur predominantly along shorelines of large lakes and rivers where invasion by woody species is also prevented by ice scouring and waves.

Table 2. Summary of Wetland Habitat Guidelines

Parameter	Guideline
Percent wetlands in the watershed and subwatersheds	Ensure no net loss of wetland area, and focus on maintaining and restoring wetland functions at a watershed and subwatershed scale based on historic reference conditions. At a minimum, the greater of (a) 10% of each major watershed and 6% of each subwatershed, or (b) 40% of the historic watershed wetland coverage, should be protected and restored.
Wetland location in the watershed	Wetlands can provide benefits anywhere in a watershed, but particular wetland functions can be achieved by rehabilitating wetlands in key locations, such as headwater areas (for groundwater discharge and recharge), floodplains and coastal wetlands. Consideration should also be given to protecting networks of isolated wetlands in both urban and rural settings.
Amount of natural vegetation adjacent to the wetland	Critical Function Zones should be established around wetlands based on knowledge of species present and their use of habitat types. Protection Zones should protect the wetland attributes from stressors. Recommended widths should consider sensitivities of the wetland and the species that depend upon it, as well as local environmental conditions (e.g., slopes, soils and drainage), vegetative structure of the Protection Zone, and nature of the changes in adjacent land uses. Stressors need to be identified and mitigated through Protection Zone design.
Wetland proximity	Wetlands that are in close proximity to each other, based on their functions, or that are in close proximity to other natural features, should be given a high priority in terms of landscape planning.
Wetland area, shape and diversity	Capture the full range of wetland types, areas and hydroperiods that occurred historically within the watershed. Swamps and marshes of sufficient size to support habitat heterogeneity are particularly important, as are extensive swamps with minimum edge and maximum interior habitat to support area-sensitive species.
Wetland restoration	Focus on restoring marshes and swamps. Restore fens under certain conditions. For effective restoration, consider local site conditions, have local sources to propagate new vegetation, and wherever possible refer to historic wetland locations or conditions. Prioritize headwater areas, floodplains and coastal wetlands as restoration locations.

2.1.1 Percent Wetlands in the Watershed and Subwatersheds

> Guideline

Ensure no net loss of wetland area, and focus on maintaining and restoring wetland functions at a watershed and subwatershed scale based on historic reference conditions.

At a minimum, the greater of (a) 10% of each major watershed and 6% of each subwatershed, or (b) 40% of the historic watershed wetland coverage, should be protected and restored.

> Rationale

All watersheds in southern Ontario currently have less wetland cover than they did prior to extensive European settlement (c. 1800), with losses exceeding 70% in many jurisdictions (Ducks Unlimited Canada 2010; Snell 1987). The guideline addresses basic minimal generic ecological functions and does not address the overall loss of unique wetland ecosystems that dominated portions of southern Ontario. Any investment in wetland restoration beyond the minimum guideline toward the historical wetland baseline will result in enhanced wetland functions and contributions to ecological goods and services. Maintenance of wetland cover across a watershed provides many ecological and hydrologic benefits. The extent of these benefits varies depending on a variety of biophysical factors including predominant landforms and soils, wetland locations, types of wetland, and predominant land uses (Flanaghan and Richardson 2010; Keddy 2010; Zedler 2003).

Historically the overall wetland coverage within the Great Lakes basin exceeded 10% (Detenbeck et al. 1999), but there was significant variability among watersheds and jurisdictions. For example, recent studies of three watersheds in the Lake Simcoe Watershed (i.e., Whites Creek, Maskinonge River and Innisfil Creek) have resulted in wetland cover estimates of 68, 52 and 59% respectively (A. Norman, OMNR, London, pers. comm. 2011), while analyses by Ducks Unlimited Canada (2010) indicate that pre-settlement wetland cover in some counties such as Peel was as low as 7.6%, and as high as 83.4% in others such as Essex.

Using historical reference points can be useful for helping to determine an appropriate level of wetland cover at the watershed or jurisdictional scale (e.g., Bedford 1999; Puric-Mladenovic and Strobl 2006). More recent historical mapping (c. 1930–1980) can also be used to help target former wetland areas that may be suitable for restoration.

One of the real challenges for maintaining key ecological and hydrological functions is in estimating the critical threshold for wetland cover in a watershed (or jurisdiction). Not surprisingly, the science on this topic shows variation among watersheds, and focuses on protection of hydrologic functions. In Wisconsin, Hey and Wickencamp (1996) examined nine watersheds and found that increasing the amount of wetland in a watershed resulted in reduced yields of water downstream, reduced flooding, higher base flows and reduced occurrence of high flows. However, these responses flattened very rapidly beyond 10% wetland cover. Zedler (2003) determined that flood abatement, water quality improvement and biodiversity support declined significantly when about 60% of the Upper Midwest's historical wetland area was drained, suggesting retention of about 40% of that area's wetland cover would continue to support those key functions. A study conducted by the University of Minnesota (Johnston et al. 1990) found that watersheds in the southern United States containing less than 10% wetlands were more susceptible to incremental losses of wetland functions than watersheds with more wetlands. This condition was found to be particularly true for flood control and suspended solids loadings.

These studies support the importance of considering watershed-specific differences in historical wetland cover, information that is readily available to jurisdictions in southern Ontario, in older as well as in updated reports (i.e., Ducks Unlimited Canada 2010; Snell 1987). In applying this guideline, current wetland cover, topography, soils and extent of impervious surfaces in a specific watershed must also be considered. In the absence of such information, the guideline of 10% cover at the watershed and 6% cover at the subwatershed scales can be used to ensure that a minimum level of wetlands are distributed throughout the watershed.

While the maintenance of wetland functions is as or more important than the maintenance of wetland area, given the limited nature of our current understanding of wetland functions, particularly at the watershed scale, wetland area serves as a useful surrogate measure.

In many watersheds in southern Ontario, particularly urbanized watersheds, it is not possible to return to historical or pre-urbanization levels of wetland cover or function because of the degree and nature of alteration that has already occurred. Nonetheless, given the known extent of wetland losses in this part of the province and the critical hydrological and ecological functions provided by these habitats, a "no net loss" approach combined with a commitment to work towards at least 40% of historical levels of coverage where it does not already exist is recommended to yield tangible benefits for communities and wildlife. The guideline can be achieved, in order of priority, through: (1) protection of extant wetlands; (2) enhancement of extant wetlands; (3) restoration of wetlands in historical locations; and (4) creation of wetlands in suitable areas.

2.1.2 Wetland Location

> Guideline

Wetlands can provide benefits anywhere in a watershed, but particular wetland functions can be achieved by rehabilitating wetlands in key locations, such as headwater areas (for groundwater discharge and recharge), floodplains and coastal wetlands. Consideration should also be given to protecting networks of isolated wetlands in both urban and rural settings.

> Rationale

Wetlands anywhere within a watershed will provide both ecological and hydrological benefits, but are increasingly being understood to provide different functions depending on their location in the watershed, as well as the characteristics of the watershed itself. The extent to which wetlands can provide water quality benefits has been linked to the biophysical characteristics of the watershed (Norton and Fisher 2000) as well as the land use context (Mitsch and Gosselink 2000; Zedler 2003).

Model-based analyses indicate that water quantity and quality benefits are derived from protection of a range of wetland sizes throughout the watershed, with small (i.e., 0.2 hectares) headwater wetlands being important for sediment removal, medium-sized (i.e., 10 hectares) mid-reach wetlands retaining significant amounts of phosphorus, and large (i.e., 250 hectares) floodplain wetlands effectively storing and attenuating long-period hydrologic flows (Cohen and Brown 2007). Earlier literature has also shown that wetlands can perform different functions depending on flow levels. For example, Johnston et al. (1990) found in their watershed-scale study in Minnesota that wetlands adjacent to streams were more effective at attenuating suspended solids, total phosphorus and ammonia during periods of high flow, and more effective at removing nitrates in periods of low flow. These results support the need for wetlands of various sizes and in various locations in the watershed, at least for provision of a more full range of water quality benefits.

In headwater areas, wetlands can provide beneficial functions. For swamps, these include protection of the quality of groundwater (discharge or recharge or both), introduction of leaves and woody debris that are essential to providing habitat for fish and macroinvertebrates downstream (Gurnell et al. 1995 cited in Detenbeck et al. 1999), and reducing the warming of streams at the source. In turn, good water-quality conditions in higher portions of watersheds are likely to benefit downstream coastal wetland ecosystems (e.g., Crosbie and Chow-Fraser 1999). Janisch et al. (2011) speak to the benefits of headwater wetlands in influencing headwater surface area processes, including improving resilience of streams following disturbances.

Further downstream, palustrine and riverine wetlands are important in reducing and asynchronizing peak flows, improving water quality, and providing habitat for aquatic invertebrates, fish and other wildlife. Richardson et al. (2011) were able to demonstrate, through a multi-phased restoration project of a stream and riparian wetland area located within the lower portion of a watershed in North Carolina, that wetlands in lower reaches associated with riparian areas can also provide significant benefits. They documented reduced downstream water pulses, nutrients, coliform bacteria and stream erosion; a substantial attenuation of nitrogen and phosphorus; and an increase in wetland plant abundance and diversity within the floodplain.

In coastal areas such as the Great Lakes, marshes are crucial habitat for fish. Wetland habitats in lakes tend to support more fish biomass than open parts of the lake, and are important in supporting fish production and species diversity (Petzold 1996; Trebitz et al. 2009). Petzold (1996) found about 60% of Lake Superior's fish biomass was associated with wetlands and concluded that these habitats are critical for the fisheries of the entire lake. This is supported by observations that changes in the amount and type of wetlands at Long Point have affected the fish assemblages populating all of Lake Erie (T. Whillans, Trent University, Peterborough, pers. comm. 2011).

Recent literature indicates that geographically isolated wetlands also provide water quality and flow regulation services, as well as maintenance of amphibian and reptile biodiversity (Comer et al. 2005). A study undertaken by Russell et al. (2002) in managed young-growth forests in the Coastal Plain of South Carolina found that isolated wetlands were focal points of amphibian and reptile richness and abundance and contributed more to regional biodiversity than would be expected based on their size and ephemeral hydrology. McKinney and Charpentier (2009) found that geographically isolated wetlands contribute to stormwater retention, flood prevention and maintenance of water quality.

Wetlands can also provide benefits that address specific objectives or problems at a watershed or more site-specific scale. Wetlands located within urban or agricultural settings act to improve water quality by retaining nutrients and sediments, and providing stormwater management (Flanagan and Richardson 2010; McKinney and Charpentier 2009). Flanagan and Richardson (2010) found that former wetland areas converted to agricultural uses were linked to higher levels of phosphorus in nearby water bodies and recommended restoration of at least some of these areas to wetland to address this issue. Some of these more localized benefits are discussed in the following section.

2.1.3 Amount of Natural Vegetation Adjacent to the Wetland

> Guideline

Critical Function Zones should be established around wetlands based on knowledge of species present and their use of habitat types.

Protection Zones should protect the wetland attributes from stressors. Recommended widths should consider sensitivities of the wetland and the species that depend upon it, as well as local environmental conditions (e.g., slopes, soils and drainage), vegetative structure of the Protection Zone, and nature of the changes in adjacent land uses. Stressors need to be identified and mitigated through Protection Zone design.

> Rationale

The amount of natural habitat that is located adjacent to wetlands can be important to the maintenance of wetland functions and attributes, particularly for wetland-dependent species that rely on these adjacent natural areas for portions of their life cycle. In cases where these adjacent natural areas form an intrinsic part of the wetland ecosystem, providing a variety of habitat functions for wetland-associated fauna that extend beyond the wetland limit, these lands can be described as Critical Function Zones (CFZs).

Beyond habitat functions, adjacent natural and semi-natural areas can also provide what are often called "buffer" functions by protecting the wetland (and its associated CFZs) from external stressors. These stressors are typically associated with human-induced changes in land use and include sedimentation, contaminants, noise, light, physical disturbances (e.g., trampling or garbage dumping), and the introduction and spread of invasive species. These adjacent areas that serve primarily a protective function are best described as Protection Zones (PZs).

Determining the appropriate amount of natural area adjacent to a wetland requires independent consideration of the CFZ and the PZ, and the functions of the two should not be confused.

Critical Function Zones and Protection Zones defined

The term Critical Function Zone (CFZ) describes non-wetland areas within which biophysical functions or attributes directly related to the wetland occur. This could, for example, be adjacent upland grassland nesting habitat for waterfowl (that use the wetland to raise their broods). The CFZ could also encompass upland nesting habitat for turtles that otherwise occupy the wetland, foraging areas for frogs and dragonflies, or nesting habitat for birds that straddle the wetland-upland ecozone (e.g., Yellow Warbler). A groundwater recharge area that is important for the function of a wetland but located in the adjacent lands could also be considered part of the CFZ. Effectively, the CFZ is a functional extension of the wetland into the upland. It is not a buffer for the wetland. See Figure 8, The Critical Function and Protection Zones.

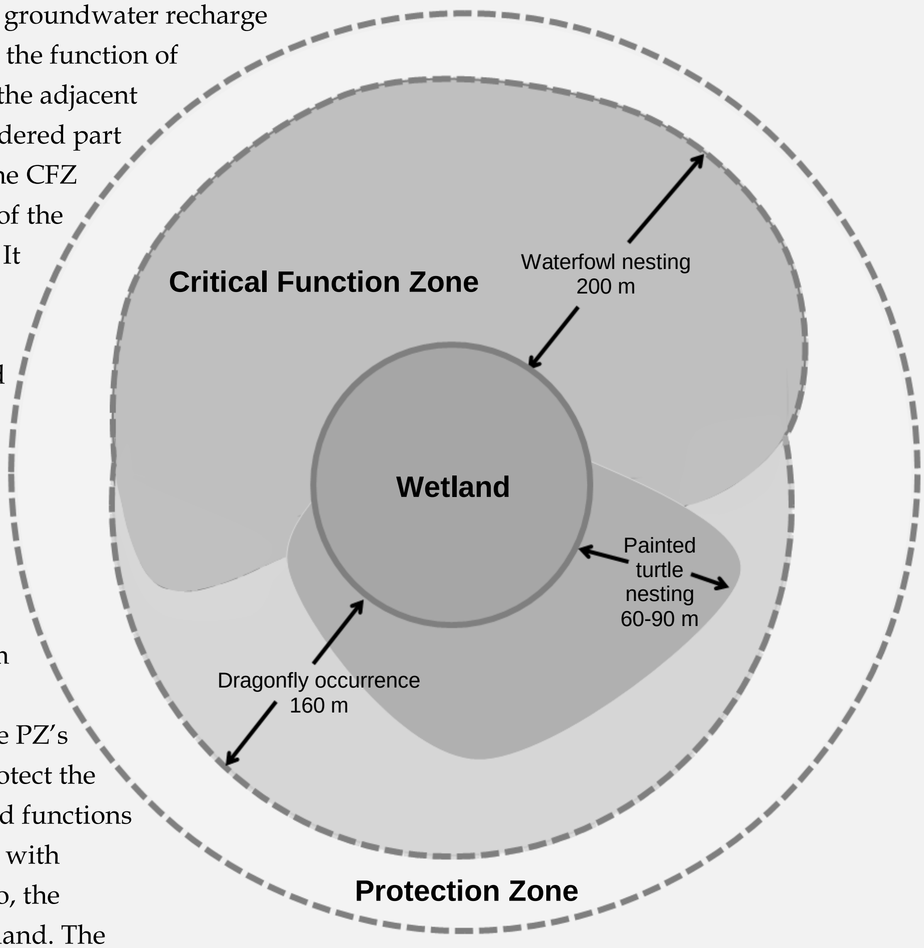

Figure 8. The Critical Function and Protection Zones

Once identified, the CFZ (along with the wetland itself) needs to be protected from adverse effects that originate from external sources by a Protection Zone (PZ). The PZ's primary function is to protect the wetland and its associated functions from stressors associated with activities in, or changes to, the lands external to the wetland. The PZ acts as a buffer in response to stressors on wetland water quality, water quantity (including the timing and degree of changes in water levels), habitat functions, or all three. PZs are analogous to filter strips and are typically vegetated areas for intercepting stormwater runoff and attenuating and transforming associated nutrients or other contaminants. They also provide physical separation from one or more stressors such as noises or visual disturbances. And they protect against direct human-associated intrusions into the wetland. Such functions are well-established in the literature (e.g., Kennedy et al. 2003; Lowrance et al. 2002; Passeport et al. 2010; Sheldon et al. 2005; Thompson et al. 2004; Woodard and Rock 1995). Depending on the nature of the stressors and the sensitivities of the wetland, alternative PZ design features such as a fence can be effective. Fundamentally, the PZ is aimed at reducing impacts on wetland functions that originate from the upland side.

A word about buffers and adjacent land

The term "buffer" is a common term used for lands adjacent to wetlands and watercourses. In *How Much Habitat is Enough?*, the term "adjacent lands" is commonly used and is meant to be interpreted literally as the lands immediately adjacent to a wetland. This is because adjacent lands may have buffer functions, important non-buffer habitat functions, or both.

These distinctions are important. For example, an area that serves a CFZ for one species may also act as a buffering PZ for another species' CFZ, or as a wetland or stream buffer.

In land use planning in Ontario, the term "adjacent lands" is used specifically to refer to "those lands contiguous to a specific natural heritage feature or area where it is likely that development or site alteration would have a negative impact on the feature or area" (MMAH 2005). These are typically prescribed as a set distance (e.g., between 50 and 120 m) from the boundary of the natural feature.

The combined CFZ and its PZ may range in total width from a few metres to hundreds of metres. Management objectives, individual characteristics of the wetland, ecological interactions with upland areas, and the source, magnitude and frequency of potential stressors and engineering options all contribute to the design of effective CFZ and PZ areas.

For CFZ determination, a good understanding of the local biophysical context, hydrologic regime and the species using the given wetland, as well as the nature and extent of their non-wetland habitat requirements, is required (e.g., Guerry and Hunter 2002; Pope et al. 2000).

Guidance regarding non-wetland habitat requirements for wildlife is increasingly available, and some of this current information is summarized in Table 3.

For wildlife, the variability in ranges of CFZs is great because of both inter- and intra-species variability in documented distances travelled for feeding and overwintering, as well as variability among breeding sites (e.g., Nichols et al. 2008). In time, more data will become available as further research (e.g., ongoing radio-telemetry studies) will lead to a more complete understanding of various species' habitat requirements.

Table 3. Selected Critical Function Zone Data for Wetlands

Species	Extent of Critical Function Zone from Wetland Edge	Reference	Notes
Reptiles			
Midland Painted Turtle	Maximum 600 m, mean 60 m; range 1 to 164 m; mean 90 m, range 1 to 621 m	Semlitsch and Bodie 2003	Review of three studies
	1 to 620 m, mean 90 m for nesting	Christens and Bider 1987	
Spotted Turtle	85 m for nesting, 54 m for dormancy	Joyal et al. 2001	Distances are mean plus standard deviation
	75 to 312 m for nesting; dormancy up to 412 m	Milam and Melvin 2001	
	Maximum 150 m; range 3 to 265 m; range 60 to 250 m	Semlitsch and Bodie 2003	Review of three studies
Blanding's Turtle	380 m for nesting, 18 m for basking and 114 m for dormancy	Joyal et al. 2001	Distances are mean plus standard deviation
	Mean 815 m, range 650 to 900 m; mean 135 m, range 2 to 1115 m; mean 168 m	Semlitsch and Bodie 2003	Review of three studies
Spiny Softshell	Mean 3 m; range 2 to 3 m; mean 0.3 m; mean 5 m	Semlitsch and Bodie 2003	Review of four studies; latter two studies single individuals
Snapping Turtle	Mean 94 m, range 38 to 141 m; mean 37 m, range 1 to 183 m; mode 25 m, maximum 100 m; mean 27 m, range 1 to 89 m	Semlitsch and Bodie 2003	Review of four studies
Wood Turtle	Mean 27 m, range 0 to 500 m; maximum 600 m; mean 60 m, maximum 200 m	Semlitsch and Bodie 2003	Review of four studies
Northern Map Turtle	Mean 2 m, range 2 to 3 m	Semlitsch and Bodie 2003	Review of one study
Eastern Musk Turtle	Mean 7 m, range 3 to 11 m	Semlitsch and Bodie 2003	Review of one study
Northern Watersnake	No adjacent lands area recommended	Attum et al. 2007	Species fairly sedentary and generally does not require upland habitat
	Maximum 6 m	Semlitsch and Bodie 2003	Review of one study

Species	Extent of Critical Function Zone from Wetland Edge	Reference	Notes
Frogs			
Western Chorus Frog	Maximum 213 m, mean 75 m	Semlitsch and Bodie 2003	Review of one study
Wood Frog	Post-breeding movements ranged from 102 m to 340 m, median 169 m	Baldwin et al. 2006a	Recommends conservation of network connected habitats
	30 m inadequate to support viable populations	Harper et al. 2008	Did not test beyond 30 m
	11 to 35 m partially mitigates timber harvest impacts	Perkins and Hunter 2006	Study is in forested landscape where the non-buffered area is logged
	40% of individuals wintered further than 100 m from breeding pond	Regosin et al. 2003	Recommends maintenance of suitable terrestrial habitat beyond 100 m
Green Frog	Mean 36 ± 25 m for foraging	Lamoureux et al. 2002	
	Mean 137 m, maximum 457 m; mean 121 m, maximum 360 m; mean 485 m , range 321 to 570 m	Semlitsch and Bodie 2003	Review of three studies
Bullfrog	Mean 406 m	Semlitsch and Bodie 2003	Review of one study
Salamanders			
Spotted Salamander	Mean maximum 106 m	Veysey et al. 2009	Salamanders used clear-cut areas to some degree
	30 m inadequate to support viable populations	Harper et al. 2008	Did not test beyond 30 m
	60% of individuals wintered further than 100 m from breeding pond	Regosin et al. 2003	Recommend maintenance of suitable terrestrial habitat beyond 100 m
	Mean 67 m, range 26 to 108 m; mean 103 m, range 15 to 200 m; mean 64 m, range 0 to 125 m; mean 150 m, range 6 to 220 m; mean 192 m, range 157 to 249 m, mean 118 m, range 15 to 210 m	Semlitsch and Bodie 2003	Review of six studies
	Range 3 to 219 m, mean 112.8 m for overwintering	Faccio 2003	Recommended "life zone" to encompass 95% of population was 175 m

Species	Extent of Critical Function Zone from Wetland Edge	Reference	Notes
Ambystoma salamanders	Mean 125 m for adults, 70 m for juveniles	Semlitsch 1998	Recommended "life zone" to encompass 90% of population was 164 m
Jefferson Salamander	Range 3 to 219 m, mean 112.8 m for overwintering	Faccio 2003	Recommended "life zone" to encompass 95% of population was 175 m
	Mean 39 m, range 22 to 108 m; mean 92 m, range 15 to 231 m; mean 252 m, range 20 to 625 m: mean 250 m	Semlitsch and Bodie 2003	Review of four studies
Blue-spotted Salamander	52% of individuals wintered more than 100 m from breeding pond	Regosin et al. 2003	Recommend maintenance of suitable terrestrial habitat beyond 100 m
Eastern Newt	No distances given	Roe and Grayson 2008	Newts are wide-ranging and active in the terrestrial habitat
	No distances given	Rinehart et al. 2009	Proximity to developed land cover essentially precluded newt occupancy
Nesting Waterfowl			
Various species in Ontario	0 to more than 400 m; 90% were within 200 m. About 20% of nests were inside or within 25 m of wetlands	Henshaw and Leadbeater 1998	Based on data from 102 nests at coastal marshes over two years. May be applicable where suitable waterfowl nesting habitat is present.
Dragonflies and Damselflies			
Various species	Dragonflies, especially Halloween Pennant, found 10 to 160 m from wetland edge	Bried and Ervin 2006	Study in Mississippi. There was a different sex balance at different distances.
	Bogs with natural habitat around them had a higher dragonfly and damselfly abundance than those with peat mining in adjacent lands	Bonifait and Villard 2010	Study in bogs in New Brunswick

Like CFZs, optimal PZ widths also vary depending on a number of site-specific factors as well as the land use context. Of primary importance is understanding the desired function(s) that the PZ is expected to perform. Key parameters that need to be considered in determining PZ widths include local hydrologic dynamics, slope, soil type(s), the vegetative composition of the buffer, and the extent and nature of the anticipated stressors (Ducros and Joyce 2003; Hawes and Smith 2005; Johnson and Buffler 2008; Polyakov et al. 2005). Examples of recommended PZ (buffer) widths for wetlands are provided in Table 4.

PZ width can also vary depending on its anticipated uses. *How Much Habitat is Enough?* encourages a shift towards the development of multicriteria evaluation approaches for PZs (or buffers) (van der Merwe and Lohrentz 2001). This approach encourages the identification and prioritization of various criteria that are selected on a site-specific basis. This could result, for example, in the encouragement of some land uses or activities within the PZs (e.g., trails), but not within the CFZs. The use of distinct "bands" within the adjacent lands area can help resolve some land use challenges when urban development is proposed close to wetlands. For example, an appropriately sized and designed PZ can accommodate trails that support opportunities for hiking and cycling, as well as nature interpretation and appreciation, or urban infrastructure such as stormwater management facilities.

Based on current knowledge, the literature increasingly indicates that the habitat requirements for wildlife tend to result in the widest and most varied CFZs (e.g., in the order of 100 metres or more). In contrast, maintaining water quality and aquatic habitat functions in streams and wetlands can often be achieved with zones of approximately only 30 metres (although there is a fair amount of site-specific variability). There are no known studies that actually test the ability of different buffer types or widths to protect wetland habitats (whether for plants or wildlife or both). Therefore, PZ recommendations related to wildlife, where provided, are typically extrapolated from measurements of impacts to various wetland species. Such recommendations may overestimate or underestimate the actual buffer widths required, and more research is required to address this knowledge gap.

Table 4. Examples of Recommended Protection Zones or Buffers to Wetlands

Stressor	Suggested Extent of Protective Zone	Reference	Notes
Sediment	6 m	Hook 2003	High attenuation rate regardless of slope (0 to 20%)
	10 to 60 m	Skagen et al. 2008	Range based on literature review
Herbicide drift from agricultural lands	Strip at edge of cultivated fields (data indicate > 6 to 9 m)	Boutin and Jobin 1998	Cites other studies suggesting 5 to 10 m
Non-point source agricultural pollutants	16.3 m grass/woody strip (riparian)	Lee et al. 2003	Removed > 97% of sediment, narrower (7 m) grass provided some benefits
Residential stormwater	15 m; 23 to 30 m on slopes greater than 12%	Woodard and Rock 1995	Groundcover type also very important
Human disturbance, landscaping (e.g., wood piles, composting)	19 to 38 m	Matlack 1993	Fencing may achieve same results in less width
Nitrate	16 to 104 m	Basnyat et al. 1999	Objective was > 90% nitrate removal
Human disturbance by watercraft	More than 80 m	Rodgers and Schwikert 2002	Based on a flush distances* of approximately 45 to 80 m for Great Lakes species (no waterfowl)
Human disturbance, recreation-related (e.g., camping, hacked trees)	67 to 130 m	Matlack 1993	
Human disturbance (on nesting Great Blue Herons)	100 m	Erwin 1989; Rodgers and Smith 1995	Flush distance* was 32 m ± 5.5 m; 40 m added to mitigate antagonistic behaviour
Urban cats	190 m	Haspel and Calhoon 1991	Measured distance predation rates on wildlife extended into adjacent natural area

* Flush distance = proximity of disturbance that will cause bird to leave nest

Useful guidance is also available from review papers that summarize data from a wide range of sources. Several review papers that have examined the available data on recommended adjacent natural areas for wetlands are summarized in Table 5. Notably, these reviews tend not to distinguish between CFZs and PZs, and instead include all adjacent land requirements into the category of "buffers." The *Planner's Guide to Wetland Buffers for Local Governments* (Nichols et al. 2008) concludes that, on average, between 30 and 91 metres of adjacent natural areas are required to support wildlife habitat functions of wetland-dependent species (with some studies recommending larger widths). The comprehensive review of temperate freshwater wetlands by Sheldon et al. (2005) concludes that there is no one effective buffer width, and recommends three possible ranges for both wildlife functions (i.e., CFZs) and buffers (i.e., PZs) combined depending on (a) the wetland's level of habitat function, and (b) the intensity of adjacent land uses ranging from 8 metres to more than 92 metres.

Table 5. Selected Reviews or Guidelines that Consider Areas of Land Adjacent to Wetlands

Reference	Recommendations for Adjacent Lands*	Notes
Brown et al. 1990	30 to 168 m for groundwater protection; 23 to 114 m for sedimentation; 98 to 223 m for wildlife habitat	Study in Florida (with a particular geology). Recommendations based on consideration of landscape conditions and information from literature review.
Castelle et al. 1994	Minimum of 15 to 30 m under most circumstances, but site-specific	Based on U.S. studies. Literature review encompasses sediment removal, nutrient removal, stormwater runoff, moderation of temperature, habitat diversity and habitat protection functions.
Lowrance et al. 2002	Range from 1 to 30 m	Based on U.S. studies. Focus on water quality functions (sedimentation and erosion, nutrient management, and pathogens and pesticides).
Norman 1996	Baseline adjacent lands area of 50 m, then subject to site-specific considerations (e.g., waterfowl production or sensitive hydrology)	Southern Ontario context. Focus on water quality functions (erosion control and reduced contamination transportation) in agricultural settings. Cautions against very narrow buffer strips.
Kennedy et al. 2003	Recommended minimums: ▪ 25 m for nutrient and pollution removal ▪ 30 m for microclimate regulation and sediment removal ▪ 50 m for detrital input and bank stabilization ▪ 100 m for wildlife habitat functions	Based on U.S. studies between 1990 and 2001

Reference	Recommendations for Adjacent Lands*	Notes
Sheldon et al. 2005	▪ 8 to 23 m for wetlands with minimal habitat functions and low-intensity land uses adjacent to the wetland ▪ 15 to 46 m for wetlands with moderate habitat functions and moderate or high-intensity land uses adjacent to the wetland ▪ 46 to 92+ m for wetlands with high habitat functions, regardless of the intensity of the land uses adjacent to the wetland	Focus on freshwater wetlands in Washington State Synthesis documents generally recommend adjacent land area widths of between 15 to 100 m, but no one width can be recommended
Bentrup 2008	No distances recommended	Recommended principles: ▪ For Critical Function Zones – larger species require larger widths, width should increase with length and in human dominated areas, and Critical Function Zones that need to function for a longer time should be wider ▪ For Protection Zones – width can be variable if variable runoff, but should be wider on steeper slopes and on soils with lower infiltration capacities
Nichols et al. 2008	▪ 9 m to 30 m for sediment and phosphorus removal ▪ 30 m to 49 m for nitrogen removal ▪ 30 m to 91 m for wildlife protection (with some studies showing more)	Review based on 50 U.S. ordinances, hundreds of scientific papers, and analyses of wetland adjacent land area performance

* Includes Critical Function Zones and Protection Zones (buffers)

Notably, even protection of wetlands and their functions through the identification and implementation of CFZs and PZs will not fully conserve the habitat functions of wetlands on a landscape scale. Implementation of these site-specific measures must be considered in the broader context of natural heritage protection on a watershed or regional scale. This is well recognized for water quality and quantity (Sheldon et al. 2005), which can be more heavily influenced by changes in land use in the broader landscape than by localized habitat protection efforts, and for wildlife such as amphibians and plants that rely on successful dispersal and migration in the broader landscape for population maintenance (Bauer et al. 2010; Keddy 2010; Semlitsch 2008).

2.1.4 Wetland Area, Shape and Diversity

> Guideline

Capture the full range of wetland types, areas and hydroperiods that occurred historically within the watershed. Swamps and marshes of sufficient size to support habitat heterogeneity are particularly important, as are extensive swamps with minimum edge and maximum interior habitat to support area-sensitive species.

> Rationale

Extensive, heterogeneous wetlands as well as less extensive, isolated wetlands both make significant contributions to supporting biodiversity at the local and watershed scales. The presence of larger, contiguous swamps and marshes (e.g., more than 30 hectares) are important for area-sensitive species such as Prothonatory Warbler and Black Tern. However, the presence of complexes of smaller, more isolated wetlands in the landscape are also important in that they provide habitat for many wetland-dependent amphibians and reptiles.

Swamps have the potential to support area-sensitive wildlife species (i.e., those that require larger areas of continuous habitat in which to be productive) or edge-intolerant species (i.e., those that prefer to use habitat away from the influence of habitat edges, also sometimes referred to as "interior" habitat species). In some watersheds with many land use pressures, treed swamps may be the only remaining significant contributors to interior-forest habitat, and the discussion on forest size and species that may be expected (see Section 2.3) applies here as well. However, treed swamps provide interior habitat for a different suite of specialist area-sensitive forest species compared to large patches of upland forest.

Larger marshes also have the ability to support area-sensitive wildlife species (Smith and Chow-Fraser 2010). Area-sensitive birds may include species such as Marsh Wren, Black Tern and Forster's Tern. Black Tern will nest in smaller marshes if larger feeding areas are located nearby. Some other species, such as Least Bittern and King Rail, occasionally occur in smaller marshes, but long-term viable populations are associated with extensive marshes.

Extensive swamps and marshes also tend to have greater habitat heterogeneity (i.e., the habitat is more varied within them), which in turn tends to support more species of wildlife (e.g., Golet et al. 2001). In marshes, this is called "interspersion" or the juxtaposition of different marsh communities. High levels of habitat interspersion (e.g., the presence of open water/submerged vegetation, emergent vegetation and in some cases shrubs) within a marsh provide higher-quality habitat for a wider variety of species than, for example, a narrow band of cattails around the shoreline. For example, some species require extensive stands of emergent plants with few or no openings (e.g., Northern Harrier), while others seem to prefer areas dominated by emergent plants but with small, isolated openings (e.g., Least Bittern). The ratio of open water/submerged vegetation to emergent vegetation and the interspersion pattern may vary considerably from year to year because marshes are dynamic systems. However, area remains a key factor, and more extensive marshes are more likely to be used as productive habitat by more species of wildlife (e.g., Attum et al. 2007; Webb et al. 2010).

Relatively isolated (i.e., not coastal or riparian), smaller wetlands can also be important for local or regional biodiversity. For example, amphibians such as Wood Frog and Spotted Salamander have been documented in wetlands ranging in size from 0.1 to 5.2 hectares (Babbitt 2005; Lehtinen and Galatowitsch 2001). These wetlands can have variable hydroperiods, may be permanently wet (i.e., year-round) or only seasonally wet (typically in the spring and part of the summer), and can be "hot spots" for some groups of amphibians, particularly when they do not support predatory or competing fish (Snodgrass et al. 2000; Werner et al. 2007). Interestingly, current research indicates that hydroperiod may be a more important factor than wetland size in determining the diversity that can be supported by isolated wetlands (Baldwin et al. 2006b; Hermann et al. 2005; Paton and Crouch 2002; Snodgrass et al. 2000).

Complexes of relatively isolated wetlands also tend to be more supportive of biodiversity than single, isolated ponds. Amphibians and reptiles are known to use multiple local ponds, sometimes over the same season (Joyal et al. 2001; Semlitsch 2008). Waterbirds are also known to use complexes of small wetlands, especially for springtime pairing and feeding, and have been documented using isolated wetlands in urban and peri-urban landscapes in London, Ontario (Pearce et al. 2007). Some birds have specifically adapted to use wetland complexes in the landscape and will readily move between them to forage (e.g., Northern Harriers, herons, dabbling ducks). This is the reason that the Ontario Wetland Evaluation System recognizes the concept of wetland complexes (OMNR 1994).

The presence of coarse woody debris in many wetland types—particularly swamps, but also riparian areas and terrestrial areas associated with wetland pockets—is important to many species. The functions of this debris include providing cover and nutrients for fish and other aquatic organisms, and providing important cover and overwintering habitat for pond-breeding amphibians that spend the bulk of their life cycle in associated uplands (Keddy 2010).

Maintenance of the full range of wetland vegetation community types that occur in a watershed is also key to sustaining biodiversity. Meyer et al. (2010) found that at Long Point on Lake Erie, while the overall abundance and diversity of birds was greater in Common Reed habitat, the abundance of marsh-nesting birds was greater in meadow marsh habitat, supporting the need to protect these types of more specialized habitats.

The role of wetland shape in supporting habitat and species diversity is difficult to discuss independently because it is so closely related to, and in the literature is often confounded with, habitat area and fragmentation in the landscape (Ewers and Didham 2006). The limited available data indicate that the optimum shape of a wetland varies by wetland type. Swamps, which are a type of forest, are better able to support area-sensitive and edge-intolerant species when they are relatively compact and regularly shaped (e.g., circular or squarish) (see Figure 13, on forest shape determining amount of core habitat, in the Forest Habitat Guidelines). However, some other wetland-dependent species require ecotonal or edge habitat (e.g., transitional areas between open water and adjacent uplands) and thrive where there is more "edge" (Attum et al. 2007; Stevens et al. 2002). Long, narrow marshes may also provide more water quality benefits since they maximize water contact with vegetation that is responsible for the uptake and transformation of many nutrients and other contaminants. The link between wildlife species diversity and abundance, and the presence of wetlands, has been made repeatedly for amphibians. Specific research on hydroperiods of seasonally inundated wetlands in forests has shown that wetlands with longer hydroperiods (but not permanent water) support a higher diversity of amphibians, irrespective of wetland size (Babbitt 2005; Baldwin et al. 2006b; Herrmann et al. 2005).

2.1.5 Wetland Proximity

> Guideline

Wetlands that are in close proximity to each other, based on their functions, or that are in close proximity to other natural features, should be given a high priority in terms of landscape planning.

> Rationale

Fragmentation of wetland habitats degrades their functions by reducing habitat for species that are less tolerant of disturbances, that require more contiguous habitat, or both, compromising the ability of individuals of a species to effectively disperse and mate with individuals from other populations, and increasing habitat for opportunistic species (such as exotic invasive species and pests). Some of these negative impacts of fragmentation can be offset, at least for some species, by maintaining concentrations of natural habitat fragments within relatively close proximity in a given landscape. This approach is well-recognized through the approach of the Ontario Ministry of Natural Resources to complexing wetlands that are within 750 metres' distance of each other. The benefits of this type of land use planning may be further enhanced by minimizing the scale and extent of built-up land uses (e.g., road size and density) in the lands within such areas.

Fragmentation of marshes within lakes can result in depletion of zooplankton and the fish species that depend on them. Even in systems where zooplankton is not a concern, small marsh patches may be ecological traps. They attract fry of many fish species as nursery habitat, but predation rates by common piscivorous (fish eating) fish such as Rock Bass may be very high.

Small marshes—especially high concentrations of small marshes in a landscape—have traditionally been conserved and restored for waterfowl production. Increasingly, the importance of adjacent natural areas, as well as proximity between patches of wetland, has become recognized for a number of other wildlife species. Attum et al. (2008) in a study conducted in Ohio and Michigan found that both Copper-bellied Watersnake and Blanding's Turtle were more likely to occur in wetlands with more surrounding forest, while Attum et al. (2007) conclude that protection of wetland complexes with a range of wetland patch sizes is needed to support Northern Watersnake foraging and mating habits. Stevens et al. (2002) found that occurrence of calling Green Frogs in restored wetlands was positively correlated with proximity to other wetlands (but provided no specific distances), while Houlahan and Findlay (2003) found in their study of 74 Ontario wetlands that amphibian richness was positively correlated with forest cover in adjacent lands, and also identified this trend in previous studies.

In terms of numerical distances for proximity, examples from the current literature are summarized in Table 6, and show a great range.

Table 6. Examples of Distances in Which Positive Relationships to Other Wetlands or Other Habitats Was Documented

Wildlife Group or Species	Distance Within Which a Positive Response Was Documented to Wetlands or Other Natural Habitats	Reference	Notes
Amphibians	Strong positive response to the proportion of wetlands within 750 to 3000 m of breeding pools	Houlahan and Findlay 2003	Studied 74 wetlands in southeastern Ontario
Amphibians	Distance to natural wetlands was an important factor in predicting amphibian diversity in the restored wetlands, particularly within the first 1000 m	Lehtinen and Galatowitsch 2001	
Turtles	Movement was concentrated within 375 m from the edge of the breeding wetland	Roe and Georges 2007	Study based in Australia
Green Frog	Occurrence increased with the percent forest cover within 1000 m of ponds	Mazerolle et al. 2005	
Spotted Salamanders	At least 100 m (range: 1.6 to 427.6 m) around seasonally inundated vernal pools was required for upland migration	Veysey et al. 2009	
Wood Frog and Spotted Salamander	Positive association with the area of upland forest within 1 km of the pond edge	Skidds et al. 2007	Studied presence of egg masses in ponds
Birds	Diversity and richness increased with the extent of forest and wetland within 500 m of the wetland edge	Mensing et al. 1998	
Birds	Presence and abundance of some species linked to the amount of wetland within a 3 km radius	Fairbairn and Dinsmore 2001	Study set in prairie marsh habitats

Additional considerations, as pointed out by Sheldon et al. (2005), include their observations that amphibians are not randomly distributed in potential habitats in the landscape but tend to be concentrated in suitable habitats that are better connected to each other, and that the presence of terrestrial habitats between wetlands can be an important factor in waterfowl distribution. Semlitsch (2008) also points out that periodic drying of temporary wetlands is an important part of their natural cycle that helps reduce amphibian predation by fish and invertebrates, and that alternate or "redundant" habitats in the landscape can provide critical refuges for amphibians when one breeding pond is disturbed or dries up.

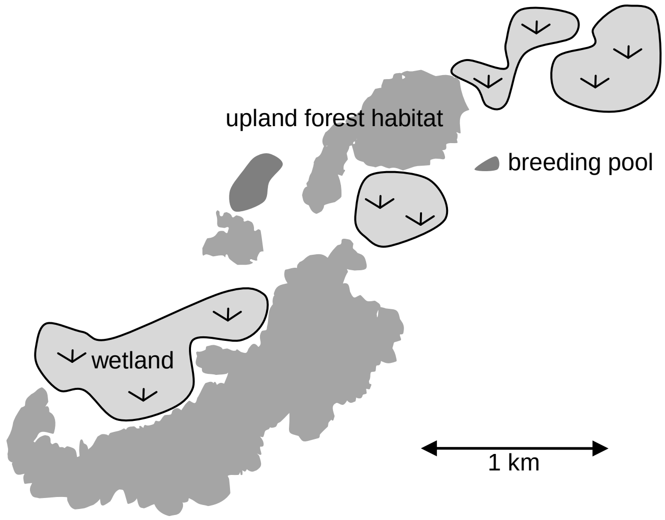

Figure 9. Wetlands in close proximity with adjacent natural areas

While most of the available research on the effects of wetland fragmentation is focused on birds and amphibians, it is intuitive that maintaining hydrologic connections between nearby wetlands (where they exist), as well as wetlands and other nearby natural areas, could also be critical to maintaining their functions.

2.1.6 Wetland Restoration

> Guideline

Focus on restoring marshes and swamps. Restore fens under certain conditions. For effective restoration, consider local site conditions, have local sources to propagate new vegetation, and wherever possible refer to historic wetland locations or conditions. Prioritize headwater areas, floodplains and coastal wetlands as restoration locations.

> Rationale

Defining restoration

The terms "rehabilitation," "creation" and "restoration" apply to distinct activities associated with habitat management and conservation. The term "restoration" in the general context of *How Much Habitat is Enough?* encompasses all these activities and refers primarily to the recreation, enhancement or improvement of habitat functions in locations or areas where the habitat had been historically present, but may also capture creation of habitats in alternate locations where conditions are suitable.

In the current land use context of southern Ontario, it is simply not possible for many watersheds and jurisdictions to return to estimated historical levels of wetland cover. Also, the ability to restore the diversity and complexity of wetlands, and their wildlife functions, remains questionable, and where possible even partial restoration of a wetland can take many years. Therefore, wetland restoration should only be considered after alternatives for protection have been examined and discarded, as a means to rectify anticipated losses, or when the objective is to increase wetland cover (see Clewell et al. 2004).

Regardless, where necessary, targeted and well-planned wetland restoration can help priority restoration areas meet water quality and wildlife habitat targets at watershed-wide and ecozone scales. Despite the limitations, restored wetlands can provide habitat for a range of species. Lehtinen and Galatowitsch (2001), in their study of amphibian colonization of 12 restored open water wetlands in Minnesota, found that restored sites were colonized by up to 75% of the species occurring in the nearby reference sites within a year. Stevens et al. (2002) found amphibian species diversity was the same between restored and reference sites, and species abundances of amphibians (Northern Leopard Frogs, Green Frogs and Spring Peepers) were higher in restored than in reference sites. Using historical data and reference sites as a starting point for setting realistic targets and identifying appropriate locations for wetland restoration is an ecologically sound approach.

Restoration by wetland type

Only two wetland types—marshes and, to a more limited extent, swamps—may be restored with some confidence.

Currently, limited information is available on the science of rehabilitating fens and bogs, and the best management strategy (as for all wetlands) is to protect them by protecting their water sources and not altering their watersheds.

In some cases, abandoned pits and quarries that are connected to the water table may offer unique opportunities for fen creation (Hough Woodland Naylor Dance Limited and Gore and Storrie Limited 1995). A current review of the state of pit and quarry rehabilitation in Ontario (Skelton Brumwell & Associates Inc. and Savanta Inc. 2009) identifies a number of locations in southern Ontario where rehabilitation of former pits and quarries has involved areas of wetland creation, including a few examples where fen vegetation has become established.

Although there is also little published research on successful reproduction of mature forested wetlands, these wetlands are considered less complex (from a restoration perspective) than bogs or fens, and are generally considered reasonable candidates for restoration given the right conditions and sufficient time for trees and tall shrubs to grow.

Marshes are considered the most readily restored type of wetland and can be at least partially functional within a few years. As a result, marsh restoration has been widely implemented. Despite this, restoration has often been unsuccessful. Available data for regulated wetland restoration from the United States, from hundreds of projects evaluated over five different states, indicate that less than a third to a quarter were considered successful in terms of area of wetland replaced for area lost. Furthermore, when specific wetland functions were compared between reference and restored wetlands, most restored or created sites had less organic matter, lower plant species diversity and structural complexity, and lower diversity of other groups of wildlife (e.g., amphibians) than their reference sites (Kettlewell et al. 2008; Sheldon et al. 2005). There have even been documented differences in levels of success between different types of marshes, with open water wetlands being most successful, shallow marsh and scrub-shrub wetland restoration being quite successful, and wet meadow restorations being relatively unsuccessful (Sheldon et al. 2005). Common problems included failure of wetland vegetation to establish throughout the site, and lower levels of vegetation and wildlife diversity compared to reference sites.

The relatively low level of documented success provides a good rationale for working towards replacement ratios of more than one to one. Another argument for wetland replacement ratio of more than one to one is presented by Gutrich and Hitzhusen (2004), who found time lags for restored wetlands to attain the floristic and soil equivalency of reference wetlands to range from 8 to 50 years.

Restoration considerations

Successful wetland restoration requires technical expertise. Key variables that need to be considered include soil conditions and fertility (including the presence of organic matter), water level fluctuations, and plant competition and structure (as determined by gradients along wetland edges). The presence of rare species is also important, not just for intrinsic value, but because they indicate the presence of rare habitat conditions that may also be valuable to other species (Keddy 2010; Keddy and Fraser 2002).

Recommendations for improving the success of wetland restoration include using a watershed approach to target and prioritize site selection, having a good understanding of the local biophysical conditions (especially hydrology, soils, slopes and potential sources for plant propagation), accepting that the restored wetland may not return to its former state or reference condition, maintaining and referring to existing hydrologic (and if possible terrestrial) linkages in the landscape, adding "ecological insurance" (e.g., try to restore more area, introduce more native diversity and incorporate a greater range of natural gradients), and where possible, undertake large-scale projects where different restoration approaches can be tested (Keddy 2010; Keddy and Fraser 2000; Keddy and Reznicek 1986; Palmer 2008; Sheldon et al. 2005; Verhoeven et al. 2008; Wagner et al. 2008; Zedler 2000). It should also be remembered that humans are not the only mammals playing a role in wetland creation: Muskrats and Beavers influence many wetland functions, and are often active participants in restoration (e.g., Gurnell 1998; Johnston and Naiman 1990; Naiman et al. 1988).

A practical process of implementing headwater wetland restoration in agricultural southern Ontario has been developed in Aylmer District of the Ontario Ministry of Natural Resources in southwestern Ontario. The process involves local biologists, drainage superintendents, landowners and others. The methodology incorporates current science, land use considerations, landowner interests and hydrological and biodiversity benefits. A guide book has been produced. This is a valuable strategy for wetland restoration that can be copied across southern Ontario (A. Norman, OMNR, London, pers. comm. 2011).

Restoration by location

Wetlands restored anywhere within a watershed will provide an array of benefits including regulation of peak water flows and increases in biodiversity, provided that they are in suitable sites. However, the scientific literature is increasingly demonstrating that restoration of wetlands in some areas can be more beneficial than in others. Some guidance for determining the best locations for wetland restoration projects is available (Almendinger 1999; Bedford 1999; DeLaney 1995; Griener and Hershner 1998).

Restoration of some major wetland areas such as Great Lakes coastal wetlands can result in extremely valuable ecological benefits; however, these projects can be technically challenging due to their size and complexity. In considering wetland location (see Section 2.1.2), restoration is best targeted to the following areas:

- Headwater wetlands, particularly swamps, should be restored where they previously existed;
- On-line or flood plain marshes and swamps should be rehabilitated or restored on second- and third-order streams;
- Rehabilitation of wetlands in lakes is a high priority because of their extreme importance to fish as well as other wildlife species; and
- Rehabilitation of wetlands in known historic locations is encouraged, where still feasible.

2.2 Riparian and Watershed Habitat Guidelines

Lands adjacent to streams and rivers are referred to as riparian. The riparian zone is an area where terrestrial and aquatic systems influence each other (Knutson and Naef 1997), and it functions as an ecotone and ecosystem (Naiman and Decamps 1997). Riparian habitat contains vegetation communities and soils with attributes of both wetland and upland areas, and provides the transition between forest and stream, hillside and valley, as well as terrestrial and aquatic ecosystems (Everett et al. 1994, as in Knutson and Naef 1997). The extent of the riparian zone is defined here as the area where vegetation may be influenced by flooding or elevated water tables (Naiman and Decamps 1997), by its related ecological functions, or both.

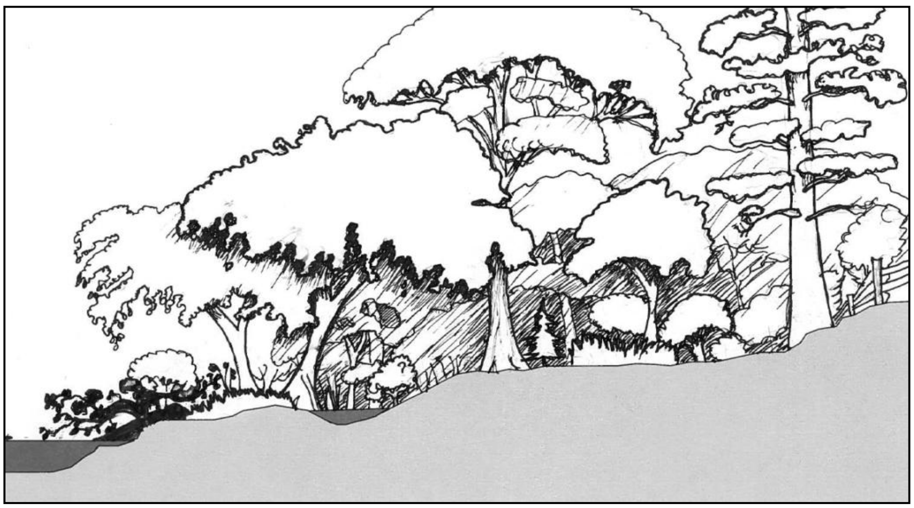

Figure 10. The riparian zone

Riparian zones provide two broad types of ecological function. They provide essential services to aquatic habitats as both a buffer between aquatic ecosystems and terrestrial systems, and as contributors of resources including woody structure, nutrients and shade. Riparian zones also provide habitat in their own right, which may be moderated or enhanced (or possibly diminished) by both the aquatic system on one side and the broader terrestrial systems on the other side. Watershed attributes beyond the riparian zone, such as land cover and land use, will also have an influence on stream habitat quality.

The habitat guidelines presented here relate to the tributaries of the Great Lakes and St. Lawrence River. The term "stream" is used here to describe any natural flowing watercourse, though unnatural (i.e., constructed or altered) watercourses may merit similar consideration depending on their function and importance within a watershed.

The focus of this section is principally on terrestrial habitat and its relation to watercourses and wetlands. As such, it does not include in-stream habitat guidance. There is a large and growing body of knowledge on in-stream habitat and hydraulic parameters that should be considered when specifically assessing stream health and considering stream rehabilitation.

The width of the riparian zone and the percent vegetated stream length guidelines directly address the amount of riparian area present to provide both direct terrestrial habitat and ecological services to aquatic habitat. It is important to recognize that this entire complex environment requires overarching general protection. This is a complex zone because riverine floodplains are species-rich systems containing ecotones at various scales between multiple habitat types (Ward et al. 1999). Fluvial processes in the form of floods and regular variations in water levels also contribute to functional and species diversity within the floodplain (Ward and Tockner 2001). In order to best address stream quality as well as terrestrial and aquatic habitat functions, the area of natural riparian vegetation should encompass the floodplain and upland transition zone or ecotone. Where there is a strong physical disjunct between the stream and upland, such as a bluff, riparian vegetation may have less direct habitat value, although it may have strong value in terms of erosion control and some habitat attributes. Widths necessary to provide effective buffering capability may also be influenced by the sensitivity of the receiving watercourse and its ability to assimilate any stressors. The width and percent vegetation guidelines represent a generic riparian zone that is applicable under the greatest range of geographic, biotic and abiotic conditions.

Impervious land cover within the broader watershed will have significant impacts on the quality of aquatic habitat within streams. These impacts may be mitigated to some degree by riparian zones. Relatively narrow riparian zones may be adequate when the broader area is in good condition (i.e., dense, native vegetation on undisturbed soils), and the adjacent land use has low to medium impact potential (i.e., parkland or low density residential). Wider riparian zones may be required to provide sufficient habitat and/or buffering functions for biologically sensitive systems, where the area is in poor condition, where soils are less permeable or highly erodible and slopes are steep, or where the adjacent land use is intense (e.g., intensive row-crop agriculture or urban centres).

Finally, measures of water quality and of fish communities provide feedback on the effectiveness of the riparian zone—in conjunction with the surrounding watershed land cover—in protecting and maintaining the aquatic environment. Fish communities may be affected by direct influences on aquatic habitat such as point source and upstream tributary inputs, or by other in-stream disturbances (human-induced or otherwise), or both. However, the quantity and quality of riparian habitat can help to directly mitigate watershed landscape effects on both water quality and aquatic life.

Contributions of the riparian zone to aquatic and terrestrial habitat

Stream size and physical characteristics associated with stream order are the products of fundamental biophysical factors (Imhoff et al. 1996; Kilgour and Stanfield 2001; Kilgour and Stanfield 2006; Stanfield and Kilgour 2006). Seelbach et al. (1997) stated that upstream catchment area, geology and slope are the primary determinants of stream size and physical conditions and, allowing for the normal distribution of plants and animals, stream biota. Modifying factors beyond land use such as in-stream barriers (including dams), channel modifications and point-source discharges will have a significant effect on stream and aquatic community qualities (Stantec 2007). Table 7 shows stream response to human alteration based on underlying conditions. Studies where the effect of adjacent vegetation has been separated from overall catchment vegetation cover show a positive relationship between the overall forest cover and stream health (Wang et al. 2006).

Small headwater systems are highly dependent upon vegetative cover for moderation of stream temperature (Broadmeadow and Nisbet 2004) and flow (Swanston 1985) as well as sediment load buffering (Dosskey et al. 2007; Dosskey et al. 2010). Additionally, these systems receive and transport large volumes of beneficial organic matter (e.g., falling leaves and insects), that are processed by fish and benthos downstream (Wipfli 2005). Headwater streams are significantly more efficient at retaining and transforming organic matter than larger streams. The retention and transformation of organic matter upstream affects downstream water quality and the survival and condition of organisms reliant on in-stream food sources (Cappelia and Fraley-McNeal 2007).

Drawing from studies on a small shaded stream, Nakano and Murakami (2001) found biomass fluxes between stream and forest accounted for 25.6 and 44.0% of the annual total energy budget of bird and fish assemblages respectively when terrestrial and aquatic ecosystems are intact. In addition, England and Rosemond (2004) suggest that relatively low levels of riparian deforestation along headwater streams can weaken terrestrial-aquatic linkages. In turn, diversity of life in first- and second-order, as well as intermittent, streams contributes to the diversity of life within the entire river and its riparian zone (Meyer et al. 2007).

Table 7. General Overview of the Sensitivity of Streams to Human Alteration of Land Cover (based on Stantec 2007)

Response Variable	Permeable Soil		Impermeable Soil	
	Small Catchment	Large Catchment	Small Catchment	Large Catchment
Mid-summer water temperature	Significant increase	Modest increase	Modest increase	Minor increase
Energy supply from inputs	Significant decrease	Minor decrease	Significant decrease	Minor decrease
Energy supply formed within	Significant increase	Minor increase	Significant increase	Minor increase
Dissolved oxygen concentration	Significant decrease	Minor decrease	Significant decrease	Minor decrease
Benthic invertebrates	Significant change to more cool and warm water, tolerant forms	Minor change to more warm water, tolerant forms	Minor change to more warm water, tolerant forms	Minor change to more warm water, tolerant forms
Fish communities	Significant change to more cool and warm water, tolerant forms	Minor change to more warm water, tolerant forms	Minor change to more warm water, tolerant forms	Minor change to more warm water, tolerant forms

From a watershed perspective, effective management practices must consider how riparian zones contribute to conditions within local streams—especially for stream orders one through three—both directly and indirectly, and how they provide terrestrial habitat in their own right. The discussions presented below provide scientifically supported guidance for the minimum habitat parameters under which riparian zones can function as aquatic buffers, wildlife corridors and in situ habitats. The definition of the riparian zone within a management context should be flexible enough to encompass these functions as well as address the need for enhanced functions to mitigate potential future impacts. Within watersheds where targets higher than these guidelines can be met and supported, they should be.

Watershed land cover and habitat health

Watershed land cover beyond the riparian zone does influence stream ecosystems; however, the relationship is difficult to quantify. Allan (2004) and other authors noted that there is an increasing recognition that human actions at a landscape scale affect stream ecosystems (Allan et al. 1997; Strayer et al. 2003; Townsend et al. 2003). This finding supports the use of detailed subwatershed studies in advising stream management decisions. Large areas of forest or other natural cover, and possibly entire catchments, may be required to maintain stream health (England and Rosemond 2004; Harding et al. 1998). However, there has been only partial success at quantifying associations between land use and effects on stream systems given covariation of human and natural influences, mechanisms operating at different scales, nonlinear stream responses, and underlying historical influences (Allan 2004). Riva-Murray et al. (2010) found that while impervious surface explained 56% of the variation in macroinvertebrate communities in the Delaware River basin, secondary land use measures explained an additional 27%. Beyond impervious cover, other potentially important aspects of watershed land cover can include measures such as urban land with tree cover, forest fragmentation (e.g., aggregation index) or aggregation of urban land use. The potential influence of these and/or other similar measures demonstrate the importance of monitoring other aspects of urbanization in addition to impervious surface.

Fitzpatrick et al. (2001) found fish communities in good condition with watershed-wide agricultural land cover up to 50% as long as riparian areas contained less than 10% agriculture. Conversely, Stanfield et al. (unpublished) found upland land cover tended not to explain any more variation in fish community composition than could be predicted by riparian buffer composition. Wang et al. (2006) found that fish community composition is more related to watershed land cover above 20% urban and greater than 70% agricultural land cover. In studies in the northeastern Ontario boreal forest and in Wyoming, 25% loss of forest cover led to effects on stream hydraulics and stream substrate composition (Buttle and Metcalfe 2000; Eaglin and Hubert 1993).

Allan (2004), in a summary of several studies, noted that there are declines in stream ecosystem health as agricultural land use increases in a watershed. Also, row crops and other more intense uses may have a greater impact on stream health than pasture. Lastly, agricultural landscapes support fewer sensitive fish and insect taxa than forested watersheds. Stream responses in terms of overall ecosystem health vary widely depending upon the study and the nature of the watershed being studied. In the same summary study, urban land use is seen as having a substantial impact on stream ecosystems, more so than agricultural land use. This difference between urban and agricultural land use was similarly observed by Snyder et al. (2003).

A number of papers examined links between aquatic ecosystems and percentage of forest cover in the surrounding landscape, and found strong connections between levels of forest cover and aquatic ecosystem health. Johnston and Schmagin (2008) found annual streamflow yields were greatest in Great Lakes basin watersheds with the highest forest cover and topographic relief. Stephenson and Morin (2009) found that forest cover at the catchment scale explained more variation in algal, invertebrate and fish biomass than any other metric. Chang (2006) cited studies that found forested watersheds normally yield streamflow of higher quality than that from other land uses. Other North American studies support the importance of forests for stream health. Cappiella et al. (2005) found healthy aquatic systems in watersheds with at least 45 to 65% forest cover. Goetz et al. (2003) looked at ratings of stream health based on indices of biotic integrity and linked 29.6% tree cover (including trees outside of natural areas) to poor stream health, 37% tree cover to fair stream health, 44.6% tree cover to good stream health and 50.6% tree cover to excellent stream health at the watershed scale. Helms et al. (2009) linked deciduous forest cover of at least 50% with higher macroinvertebrate species richness. Stephenson and Morin (2009) reported that when forest cover was less than 50%, algal biomass was relatively high (but patterns were variable), and that fish biomass began to decline where forest cover was less than 45% at the reach scale.

Moreover, higher percentages of porous land cover across the watershed, such as forest, wetland and meadow, will have a positive effect on stream ecology on the basis that they are not impervious surfaces.

Forest cover between 40 to 50% has a positive effect on stream ecosystem health, which supports the percentage forest cover guideline in the Forest section of this report (see Section 2.3).

Table 8. Summary of Riparian and Watershed Habitat Guidelines

Parameter	Guideline
Width of natural vegetation adjacent to stream	Both sides of streams should have a minimum 30-metre-wide naturally vegetated riparian area to provide and protect aquatic habitat. The provision of highly functional wildlife habitat may require total vegetated riparian widths greater than 30 metres.
Percent of stream length naturally vegetated	75% of stream length should be naturally vegetated.
Percent of an urbanizing watershed that is impervious	Urbanizing watersheds should maintain less than 10% impervious land cover in order to preserve the abundance and biodiversity of aquatic species. Significant impairment in stream water quality and quantity is highly likely above 10% impervious land cover and can often begin before this threshold is reached. In urban systems that are already degraded, a second threshold is likely reached at the 25 to 30% level.

2.2.1 Width of Natural Vegetation Adjacent to Stream

> Guideline

Both sides of streams should have a minimum 30-metre-wide naturally vegetated riparian area to provide and protect aquatic habitat. The provision of highly functional wildlife habitat may require total vegetated riparian widths greater than 30 metres.

> Rationale

Vegetation communities within the riparian zone can directly influence aquatic habitat and affect water quality for aquatic life. These functions include moderation of temperature through the provision of shade, filtration of sediments and nutrients, provision of food inputs through organic debris and leaf litter, and contribution to physical habitat in terms of fallen woody material. Vegetated riparian zones also serve as terrestrial habitat and corridors for wildlife as well as places where terrestrial and aquatic food webs interconnect. These diverse functions can be interactive or independent and will vary with watershed context. For example, the ability of riparian vegetation to moderate water temperature may decline with increasing stream width and volume, but it may still provide terrestrial habitat.

The riparian zone width requiring maintenance or protection may vary depending on the size (order) of the stream, the steepness of the banks, and the specific management concerns of the local system (USDA 2007). Kennedy et al. (2003) provide detailed tables of riparian widths associated with specific conservation concerns and broader landscape considerations. The 30-metre width guideline provided here is a minimum general approximation intended to capture processes and functions typical of the active riparian zone of a floodplain and the floodplain-to-upland transition with respect to ecological services provided to aquatic habitat.

The riparian width guidelines do not directly include transition buffers beyond the riparian zone, but transition buffers should be considered in managing the riparian zone and from an ecosystem management approach. The type of vegetation and other site-specific conditions beyond the immediate riparian zone may be of particular importance in the management of urban watersheds, as urban development entirely changes the characteristic of surface flow that laterally enters the riparian. The effects of vegetation and land cover beyond the riparian zone on stream aquatic habitat are discussed below and in the following section. Also, while adjacent vegetation should be maintained next to lakes for similar reasons as for streams, this guideline was not developed specifically for lakes.

In terms of buffering and habitat functions, there are parallels with PZs as discussed in Section 2.1.3, Amount of Natural Vegetation Adjacent to the Wetland, but with a few key differences. Principally, the 30-metre riparian adjacent vegetation guideline is not based on a species- or function-specific need but reflects a general threshold distance for aquatic health and riparian functions. Also, the 30-metre width is meant to capture a variety of protection and habitat functions. And some of the riparian habitat functions, such as wildlife corridors, do not reflect upland habitat needs of aquatic species but the needs of upland species that utilize the riparian system or stream. While the PZ, and to a degree CFZ, concepts can be applied to riparian systems with careful consideration under certain circumstances, it is suggested that, at a minimum, 30 metres of natural vegetation be maintained next to streams.

Knutson and Naef (1997) reviewed numerous published sources on varying riparian widths and the effect on stream health. Reported widths ranged from 3 to 200 metres but with prevalence in the 23 to 60 metre range (all of these are applied to both sides of the stream from the edge of the watercourse inland). They concluded by recommending that fish-bearing streams have buffers of either 46 metres or 61 metres (for streams less than or greater than 1.5 metres' width respectively), extending to 76 metres for shorelines or streams of state-wide significance. In a later review that reported on studies from across North America and Europe, Broadmeadow and Nisbett (2004) cited ranges of 15 to 70 metres for stream temperature moderation, 15 to 100 metres for sediment removal and control, and 27 to 100 metres for woody debris and leaf litter supply. A partial removal or loss of some riparian trees, however, may not necessarily impair some riparian buffer functions. For example, Wilkerson et al. (2006) found only a 60% canopy cover is required to maintain effective temperature control.

In terms of landscape context, Wang et al. (2003) found that land cover within 30 metres of streams in Wisconsin and Minnesota explained more variation in fish assemblages than land covers beyond 30 metres. Frimpong et al. (2005) found that buffers that were 30 metres wide and 600 metres long were the best predictors of the composition of fish communities within Indiana streams. However, watershed land cover, and more specifically the ratio of natural cover (especially, but not necessarily forest cover) to human-dominated impervious land covers, will change the effectiveness of riparian buffers. Roy et al. (2007) found that 30-metre forested buffers protected fish stream communities, but with diminishing returns above 15% urban land cover within the catchment. The effects of catchment land cover are important and are further discussed in subsequent sections.

In reviews by Castelle et al. (1994) and O'Laughlin and Belt (1995), riparian zone widths of 3 to 200 metres were found to be effective for bank stabilization and sediment erosion control, depending on site-specific conditions. The Castelle et al. (1994) review specifically looked at different riparian widths' effects on sediment removal. The relationship between width and sediment removal was non-linear, with disproportionately wider riparian strips required for relatively small improvements in sediment removal. For example, in one test case, widths of 30.5 metres removed 90% of sediments on a 2% slope; however, a width of 60 metres was necessary to remove 95% of sediments on the same slope. On steeper slopes, two other studies found that widths of 60 metres were effective in removing greater than 80% of sediments (Castelle et al. 1994). The frequency and intensity of sediment inputs are important criteria for the effectiveness of the riparian zone in mitigating the effects of sediment inputs.

Castelle and Johnson (2000) found that most contributions to aquatic habitat are realized in the first 5 to 30 metres of the vegetated riparian zone (with rooted vegetation contributing to sediment filtering and trees contributing wood structure and shading). In heavy rain events, riparian sediment filtering may be augmented through the use of additional grassy strip areas upland of the riparian zone, which can reduce concentrated surface flows (Knight et al. 2010). Knight et al. (2010) observed forest buffers of 17 to 18 metres remained unbreached (i.e., did not develop lateral erosion channels) by concentrated flow events with the addition of 18 to 22 metres of grassy buffer beyond a treed riparian area.

Adjacent lands with established vegetation are fairly efficient at removing excess nutrients from surface runoff. In some studies, areas with widths as narrow as 4.6 metres have been 90% effective in removing nitrogen and phosphorus, but most areas require a minimum of 10 to 15 metres. A 30-metre-wide adjacent land area along a stream next to logging operations greatly reduced nutrient levels to better than drinking water standards. Wooded riparian lands in Maryland removed 80% of excess phosphorus and 89% of excess nitrogen, mostly within the first 19 metres. Lee et al. (2003) found that more than 97% of sediment and 80 to 90% of key nutrients could be removed with 16.3 metres of mature grass/woody riparian adjacent land area.

Riparian zones as wildlife habitat

Riparian systems can provide important wildlife habitat. Habitat may be valuable for its intrinsic values, for example as forested habitat for breeding birds or as habitat for flora or as linear features providing connectivity for terrestrial wildlife movement (Knutson and Naef 1997), rather than any particular relationship to the riparian zone itself. Corridor and habitat widths for mammals, reptiles and amphibians are often dependent on the requirements of individual species, and these are discussed elsewhere in this document. However, it is worth noting that widths to address ecological concerns are much wider than those recommended for water quality concerns (Fischer 2000; Fischer and Fischenich 2000). As the width increases, factors such as overall habitat heterogeneity become important and the habitat requirements of species exceed the area of the riparian zone.

Riparian areas, specifically due to their association with water, provide core habitat areas for many herpetile (reptile and amphibian) species. Semlitsch and Brodie (2003) suggest that the 15 to 30 metres of adjacent land often prescribed as a buffer to protect wetland species is inadequate for amphibians and reptiles, as riparian areas are not buffers per se but rather are core terrestrial habitats used by these species. The minimum suggested riparian zone requirements for some herpetiles ranged from 127 to 205 metres.

There is a wide range of suggested appropriate riparian widths based on function, with the published range in the literature varying from a few metres to over 100 metres, depending on the study and level of representation and confidence (e.g., 95% occurrence within 175 metres). The recommended 30-metre width is supported in the literature as a general guideline minimum for many riparian systems. The 30-metre guideline may provide for basic terrestrial habitat; however, a greater width may be required to provide for a highly functional wildlife habitat.

2.2.2 Percent of Stream Length Naturally Vegetated

> Guideline

Seventy-five percent (75%) of stream length should be naturally vegetated.

> Rationale

This guideline focuses on the cumulative effect of the riparian zone on stream habitat, and water quality as related to stream habitat. As described, riparian zones contribute to stream habitats in many ways. At the local scale, natural but otherwise open landscapes (i.e., not forested) require a forested riparian zone of at least 150 metres in order to generate substrate habitat capable of supporting benthic communities other than those tolerant to human disturbance (Wooster and DeBano 2006).

A Toronto-area study found stream degradation occurred when riparian vegetation amounted to less than 75% cover along the length of first- to third-order streams (Steedman 1987). Alternatively, in a field test of this guideline, the Toronto and Region Conservation Authority commented that there are many cold water streams that have less than 75%, or even less than 50%, vegetated riparian habitats, pointing to an ability for some streams to resist degradation below the 75% threshold.

Related comments were provided by Gartner Lee Limited (1997a) in a field test in Hogg Creek located in the Severn Sound Area of Concern, Ontario. In Hogg Creek, only 43% of the first- to third-order streams had vegetated riparian zones. Several tributaries of the main branch of Hogg Creek exhibited cold water characteristics that seemed to relate to a high ratio of baseflow (approximately 47%) as a percentage of average annual discharge per square kilometre. Gartner Lee Limited (1997b) also noted that the presence of cold water streams is heavily dependent on the geological characteristics of the area.

Riparian vegetation provides proportionately greater benefits to stream aquatic habitat along the headwaters of streams. There is growing literature on the interaction between stream and adjacent terrestrial communities (Iwata et al. 2010; Nakano and Murakami 2001). From a watershed perspective, planting vegetation along smaller systems (i.e., less than third-order) will produce greater aquatic habitat benefits than planting along higher-order rivers, although both provide benefits. Vegetation along a smaller stream may have a greater potential to provide sufficient cover in order to lower the summer maximum stream temperatures than along the banks of a large river, but more importantly, aquatic/terrestrial biomass exchange may be more balanced in smaller streams with abundant adjacent cover. Given that the form and function of headwater streams are strongly influenced by the character of adjacent lands, a lack of adjacent natural vegetation can result in dramatic alterations in flow and sediment regimes.

Riparian vegetation, however, will still be of great value along larger rivers for species such as waterfowl, reptiles and amphibians, as well as mammals such as North American River Otter, American Mink and Beaver. In this context, there is a similarity to wetland CFZs (see Section 2.1.3): within riparian zones, there will be species-specific areas of adjacent land required to complete life cycles for species that spend their life cycle in both aquatic and terrestrial habitats.

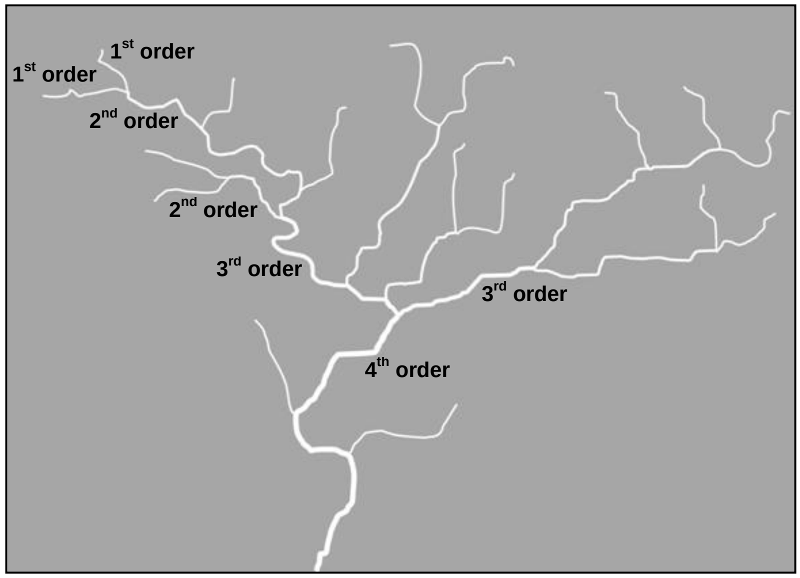

Figure 11. Stream order, showing extent of first- to third-order streams

Lower-order streams comprise the majority of stream systems in the lower Great Lakes basin. For example, approximately 75% of the length of the Credit River system is comprised of first- to third-order streams, 83.5% of Duffins Creek is first- to third-order streams, and 75% of Carruthers Creek is first- to third-order streams (TRCA 2012; CVC 2012 pers. comm.).

The 75% guideline is presented as a minimum, given the literature and the fact that headwaters in southern Ontario tend to comprise approximately 75% of stream lengths. Effective management approaches should consider local watershed conditions (land cover) when applying this guideline, as discussed in the following section.

2.2.3 Percent of an Urbanizing Watershed that Is Impervious

> Guideline

Urbanizing watersheds should maintain less than 10% impervious land cover in order to preserve the abundance and biodiversity of aquatic species. Significant impairment in stream water quality and quantity is highly likely above 10% impervious land cover and can often begin before this threshold is reached. In urban systems that are already degraded, a second threshold is likely reached at the 25 to 30% level.

> Rationale

As part of the hydrologic cycle, water falling as precipitation will soak into the ground replenishing ground water and deeper aquifers (infiltration), or will flow above ground as surface water. Both ground and surface water may enter streams, rivers and lakes, eventually flowing into oceans. The base flow of a stream relies on ground water. During a precipitation event, if precipitation exceeds the infiltration capacity, water runs along the surface, either infiltrating elsewhere or entering streams directly. Total surface runoff flowing directly into streams increases with impervious land surfaces, such as pavement and concrete, or even with reduced infiltration, such as on compacted soils (Stanfield and Jackson 2011).

The replacement of natural vegetation with impervious surfaces contributes to altered runoff processes within urban watersheds (Booth 1991; Booth 2000; Booth et al. 1997; Knutson and Naef 1997). The loss of fish and wildlife habitat, along with channel erosion and downstream flooding, are primary components of a stream system decline that result from high impervious levels within a watershed (Booth 1997; Booth 2000; Knutson and Naef 1997). While riparian zones can reduce surface runoff through increased infiltration (Bharati et al. 2002), there is a limit to their effectiveness in watershed conservation. In a review of the limits of best management practices, riparian buffers ceased to provide any protection with respect to nutrient loadings and turbidity, regardless of riparian width and percent length stream coverage, when impervious coverage ranged from 12 to 45% (Brabec 2009). More importantly, even with the full retention of complete natural riparian coverage, measurable degradation in stream quality was observed at watershed impervious coverage of just 7 to 10%.

The effects of natural vegetation loss to impervious surfaces are often permanent (Booth 1991), and in this regard implementing mitigation efforts after impervious surfaces are established is largely unsuccessful (Booth et al. 1997). Stream biodiversity may continue to be negatively affected by landscape scale disturbances for decades despite improvements to riparian zones and associated buffering functions (Harding et al. 1998).

The debate on identifying reasonable thresholds for impervious surfaces within a watershed began in 1979. In his pivotal paper, Klein (1979) reported that impairment in stream quality is first noted at 10 to 12% impervious cover and becomes severely impaired at 30% watershed imperviousness. In a subsequent literature review, the Stormwater Manager's Resource Center proposed that two thresholds exist within urbanized watersheds: at 10% imperviousness, certain stream quality parameters will be affected, and at 25 to 30% impervious cover, stream quality will consistently shift to a degraded condition (Schueler 2003).

As watershed imperviousness surpassed 10%, Booth (1991) and Booth et al. (1994) found that there was a rapid decline in fish habitat and channel stability of riparian zones. In addition, Booth (1991) stated that urban development both magnifies peak discharges and creates new peak runoff events.

Schueler (1994) reported on a number of studies that relate imperviousness to runoff characteristics, stream morphology, water quality, pollutant loading, stream warming, as well as aquatic biodiversity. In his review, he suggested that impervious land use should remain below 10% as a guideline to protect streams. Snodgrass (1992) reported that water quality became degraded when hard surfaces from development (e.g., housing, roads) reached 15 to 25% of the watershed. Contemporary stormwater management could not prevent stream-quality impairment in the study provided by Snodgrass (1992). In the past two decades, stormwater best management practices have evolved considerably. However, the primary focus of control of peak flow rate and the reduction of suspended solids has not mitigated the widespread and cumulative hydrologic modification to both streams and the broader watershed (CVC and TRCA 2010).

Various indicators of aquatic macroinvertebrate community health are widely used as relationship indicators between watershed imperviousness and aquatic systems. The thresholds presented below are taken from the Stormwater Manager's Resource Center review (Schueler 2003). As impervious cover increased from 8 to 9% within a watershed, there was significant decline in wetland aquatic macroinvertebrate health (Hicks and Larson 1997). When the percentage of total impervious surfaces surpassed 5 to 10% of a watershed landscape, there was a rapid decline in biological stream indicators (May et al. 1997). At a study conducted in Washington, D.C., a significant decline in the diversity of aquatic insects was noted at 10% impervious cover (Anacostia Restoration Team 1992). Furthermore, the density and diversity of wetland plants, amphibians and fish are also impaired as watershed imperviousness exceeds 10% (Limburg and Schmidt 1990; Taylor et al. 1995).

The most commonly chosen threshold for impervious surfaces is 10% of the land cover within a watershed (Booth 2000). This value is proposed as a defensible minimum standard for a guideline. Not every watershed, though, will respond uniformly or as anticipated to proposed impervious surface thresholds. While some watersheds might maintain integrity at 10%, as imperviousness moves toward 10%, the fish community may already have simplified, with an increase in generalist fish such as carp and perch (Wang et al. 1997; Weaver 1991). Effects on aquatic life can be seen between 5 to 8% imperviousness (Horner et al. 1997; May et al. 1997; Shaver and Maxted 1995; Yoder and Rankin 1995), and in some cases even lower (King et al. 2011; Stranko et al. 2008; Utz et al. 2009; Utz et al. 2010). Wang and Kanehl (2003) found that high-quality macroinvertebrate communities were possible in cold water streams if impervious land cover constituted less than 7% of the watershed area, but that low quality index scores were inevitable above 10% imperviousness. Imperviousness levels between 7 and 10% represented a threshold urban development zone where minor changes in urbanization could result in major changes in cold-water stream macroinvertebrate communities.

Stranko et al. (2008) observed Brook Trout presence in small suburban and urban streams to be strongly positively correlated with watershed forest cover and their absence from cold water streams with as little as 4% watershed impervious cover. Effects on stream channel geomorphology can be seen at as low as 2% (Morisawa and LaFlure 1979) and 4% (Dunne and Leopold 1978). These studies suggest that even a 10% guideline may be too high to adequately protect an urbanizing watershed. Moreover, it is clear that a 10% limit (or any suggested limit) does not necessarily represent a tipping point below which streams will not have an impact from impervious cover. For example, negative linear responses for Brook Trout (Stanfield et al. 2006) still occur below this threshold. Guideline values of 5 to 7% impervious cover provide a more conservative limit for urbanizing watersheds, though they may be difficult to obtain and maintain. For urban watersheds that have, to date, exceeded the 10% impervious surface guideline, a second threshold of 25 to 30% or less impervious surfaces is suggested.

In southern Ontario, impervious surfaces are often associated with particular land uses, and the relationships between land cover and stream water quality depends, in large measure, on how land cover is classified (Stantec 2007). While urban lands are generally more impervious than agricultural lands, there can be significant differences in permeability within these categories (Stantec 2007). Industrial areas with large parking lots can have upwards of 90% imperviousness while estate lot residential areas can have values between 5 and 10% imperviousness (Arnold and Gibbons 1996; Prisloe et al. 2001). Imperviousness also varies with the agricultural type and intensity level. In a New Zealand study addressing land use and overall stream health, Harding et al. (1999) suggest that measures of agricultural intensity rather than percent cover of agricultural land use may provide more useful land management thresholds.

In relatively undeveloped rural watersheds, stream base flow is dictated by underlying soils and geologic conditions that influence the amount of ground water discharge. Within urbanizing watersheds, however, careful planning may mitigate some of the effects of impervious surfaces. Extreme peak flows typical of urban environments can be reduced through minimizing hard surfaces. Booth et al. (1997) suggested that by using construction methods and products such as permeable pavements that allow for infiltration of water, surface area covered by otherwise impermeable constructed surfaces (rooftops, pavement, compacted soils) can be reduced.

Imperviousness provides a surrogate measure of a variety of stream impacts associated with watershed development and land use conversion. In determining conservation policies, however, it is important to understand that impervious cover should not only be considered as the percent of a watershed or subwatershed that is paved or developed. There is emerging literature on the intensity and nature of land uses and the relationship to aquatic habitat and water quality, such as the work by Coles et al. (2010) to develop an urban intensity index related to stream ecological condition. In this study, an urban intensity value was derived using 24 landscape variables and measured at 30 sites representing different levels of watershed development. From this, a gradient of urban intensity was developed that allowed estimates of expected stream condition based on level of urbanization.

Stanfield and Kilgour (in review) examine a land disturbance index in southern Ontario as a more inclusive measure of disturbance that measures the cumulative development within a watershed. Beyond the simple percent impervious guideline value, effective conservation planning and policy should consider where the imperviousness is located in a watershed. If headwater areas have minimal impervious cover, and urban areas with high imperviousness are located only in the higher-order sections of the watershed, streams may have a better chance of maintaining integrity.

Future changes can be anticipated both in the way in which imperviousness is calculated and the manner in which intensity of land uses is factored into this assessment. In the interim, the guidelines provided make the best use of the science surrounding this subject.

2.2.4 Additional Riparian and Watershed Considerations

Physical and chemical water quality parameters

Physical and chemical measures of water quality should be within federal and provincial guidelines.

Measures of water quality such as total suspended solids, pH, oxygen, and concentrations of nutrients, metals and other contaminants are important in monitoring the health of streams. Water quality parameters are affected by influences on aquatic habitat such as point source and non-point source upstream tributary inputs, in-stream disturbances (anthropogenic or otherwise), or both, and by the conditions in the broader watershed such as levels of impervious cover (as discussed above). The quantity and quality of riparian habitat can mitigate watershed landscape effects on water quality (Brabec 2009).

In Canada, the Canadian Council of Ministers of the Environment (CCME) Water Quality Task Group develops federal guidelines for the protection of aquatic life covering water quality, sediments and tissue. Canadian Environmental Quality Guidelines provide chemical-specific guideline fact sheets indicating environmental limits for over 200 different chemicals and water quality parameters (CCME 1999). Each fact sheet summarizes key scientific information and the rationale for the limit. They also provide detailed implementation advice. The Canadian Environmental Quality Guidelines and Summary Tables are available online at http://st-ts.ccme.ca.

Beyond individual parameter guidelines, the CCME has also developed a spreadsheet tool that summarizes multiple water, sediment and soil quality variables into a single measure of overall water, sediment or soil quality. The Water Quality Index was developed to provide a standard metric for jurisdictions to track and report water quality information. It is available, along with associated support material, at www.ccme.ca/ourwork/water.html?category_id=102.

In southern Ontario, provincial water quality objectives (PWQOs) also provide standard guidelines for physical and chemical water quality parameters for the protection of the health of aquatic life. PWQOs provide guidance in making water quality management decisions and should be considered in the management of riparian zones and broader watersheds, given their influence on streams and other aquatic habitat. The Ontario Ministry of the Environment provides more than 240 provincial water quality objectives in its publication *Water Management, Policies, Guidelines: Provincial Water Quality Objectives of the Ministry of the Environment*, also known as the "Blue Book." It is available on the Ministry's website at www.ene.gov.on.ca/environment/en/resources/STD01_076352.html.

2.2.5 Fish Community Targets

Fish communities are a product of stream and watershed characteristics, and there are various guides available to measure the health of aquatic habitats and to establish fish community targets.

In setting fish community targets, there are two basic reference points:

- The fundamental or underlying characteristics of the stream and watershed or subwatershed (e.g., drainage area, geology, stream substrate, gradient, flow regime, etc.), and the makeup of historical fish communities; and
- The biota currently present in the stream (i.e., fish communities and other aquatic communities), the existing aquatic habitat conditions, and the factors that presently impact the system and their relative magnitudes.

The fundamental characteristics of the stream and watershed dictate the limits and potential of the stream system. The historic condition provides a direction for rehabilitation. The existing conditions indicate how far the system is from being healthy or at least resembling historic conditions.

Beyond these fundamental reference points and characteristics, there are a variety of approaches to restore stream environments and manage fisheries. In order to develop locally relevant fish community targets, it is advisable to contact agencies and organizations working in your local watershed, such as the Ontario Ministry of Natural Resources, Conservation Authorities or non-government groups.

2.3 Forest Habitat Guidelines

In this document, the term "forest" includes all treed communities (where trees are generally 6 metres or more in height) with a canopy cover of at least 35%, and more typically 60% or greater. This includes both upland forests and swamps as well as plantations. It generally does not include orchards or tree farms.

Prior to European settlement, forest was the predominant habitat across the Mixedwood Plains. And today, if human influence and disturbance were to cease, it would be the land cover that would be most likely to naturally re-establish across most of the ecozone. Many of the types of wildlife that are currently found in the Mixedwood Plains, and the niches they occupy, are a legacy of this past forest matrix. The remnants of this vast forest now exist in a fragmented state with patches of various sizes distributed across the settled landscape, with higher levels of forest cover occurring along the northern edges of the ecozone and associated with features such as the Niagara Escarpment and Frontenac Axis. The forest legacy, in terms of species richness, ecological functions and ecosystem complexity is still evident in these patches and regional forest matrices. These ecological features are in addition to the previously discussed influence forests have on water quality and stream hydrology (see Section 2.2, on the riparian zone), and include reducing soil erosion, producing oxygen, storing carbon and many other ecological services that are essential not only for wildlife but for human well-being.

Many flora and fauna species are obligate users of forested habitats—that is, they cannot survive without forested habitats. Structurally diverse (compared to many other habitats), forests provide a great many habitat niches that are in turn occupied by a great diversity of species. They provide food, water and shelter for these species—whether they are breeding and more or less resident, or using forest cover to assist in their movements across the landscape. This diversity of species includes many that are considered to be species at risk.

From a wildlife perspective, there is increasing evidence that the total forest cover in a given area is a major predictor of the persistence and size of bird populations, and it is possible or perhaps likely that this pattern extends to other flora and fauna groups. The pattern of distribution of forest cover, the shape, area and juxtaposition of remaining forest patches, and the quality of forest cover also play major roles in determining how valuable forests will be to wildlife and people alike.

The following series of forest habitat guidelines relate to the amount of forest cover, the area and shape of forest patches, the configuration of forest patches, connectivity between forest patches, and forest quality, as summarized in Table 9.

Table 9. Summary of Forest Habitat Guidelines

Parameter	Guideline
Percent forest cover	30% forest cover at the watershed scale is the minimum forest cover threshold. This equates to a high-risk approach that may only support less than one half of the potential species richness, and marginally healthy aquatic systems; 40% forest cover at the watershed scale equates to a medium-risk approach that is likely to support more than one half of the potential species richness, and moderately healthy aquatic systems; 50% forest cover or more at the watershed scale equates to a low-risk approach that is likely to support most of the potential species, and healthy aquatic systems.
Area of largest forest patch	A watershed or other land unit should have at least one, and preferably several, 200-hectare forest patches (measured as forest area that is more than 100 metres from an edge).
Forest shape	To be of maximum use to species such as forest-breeding birds that are intolerant of edge habitat, forest patches should be circular or square in shape.
Percent of watershed that is forest cover 100 m from forest edge	The proportion of the watershed that is forest cover and 100 metres or further from the forest edge should be greater than 10%.
Proximity to other forested patches	To be of maximum use to species such as forest birds and other wildlife that require large areas of forest habitat, forest patches should be within two kilometres of one another or other supporting habitat features. "Big Woods" areas, representing concentrations of smaller forest patches as well as larger forest patches, should be a cornerstone of protection and enhancement within each watershed or land unit.
Fragmented landscapes and the role of corridors	Connectivity width will vary depending on the objectives of the project and the attributes of the forest nodes that will be connected. Corridors designed to facilitate species movement should be a minimum of 50 to 100 metres in width. Corridors designed to accommodate breeding habitat for specialist species need to meet the habitat requirements of those target species and account for the effects of the intervening lands (the matrix).
Forest quality – species composition and age structure	Watershed forest cover should be representative of the full diversity of naturally occurring forest communities found within the ecoregion. This should include components of mature and old growth forest.

2.3.1 Percent Forest Cover

> Guideline

30% forest cover at the watershed scale is the minimum forest cover threshold. This equates to a high-risk approach that may only support less than one half of the potential species richness, and marginally healthy aquatic systems;

40% forest cover at the watershed scale equates to a medium-risk approach that is likely to support more than one half of the potential species richness, and moderately healthy aquatic systems;

50% forest cover or more at the watershed scale equates to a low-risk approach that is likely to support most of the potential species, and healthy aquatic systems.

> Rationale

Forest habitat thresholds: proceed with caution…

There is tremendous interest in minimum threshold levels for forest cover (and other habitat types) required to support healthy levels of native flora and fauna. Such thresholds can facilitate natural heritage planning and help support initiatives to protect and expand forest cover. However, given the current data gaps in the site-specific and landscape-scale habitat requirements of different species and groups of species, the science that is currently available to support such thresholds is limited. While there are several studies that examine forest cover thresholds, these studies acknowledge that the research has been conducted for relatively few species and taxonomic groups (primarily birds), and is rarely undertaken for long-enough periods of time, at large-enough scales, or with sufficient replicates to make the findings robust (Betts and Villard 2009; Lindemayer et al. 2005; Price et al. 2007; Rompré et al. 2010). Therefore, the values provided here should be viewed as generalized guidance based on the available science.

…but consider risk

The current science generally supports minimum forest habitat requirements between 30 and 50%, with some limited evidence that the upper limit may be even higher. Price et al. (2007) suggests a risk-based approach to recommendations for minimum habitat requirements that takes the current science, as well as the known uncertainties, into account. Risk-based thresholds provide a more nuanced approach, and the approach has been applied to the forest cover guideline (see Section 2.3.1). This makes intuitive sense, as there is seldom a clear-cut answer to a particular problem in ecology. There is potential to adopt this approach for further habitat guidance within the Mixedwood Plains.

Despite new evidence about the influence of the matrix, current research continues to show that overall levels of forest cover are much more important determinants of long-term species persistence than the nature of the intervening habitats, particularly for forest birds (Price et al. 2007; Watling and Donnelly 2006). This is increasingly evident as levels of forest cover increase in a given landscape. For example, Donnelly and Marzluff (2004) reported that in a landscape with about 60% forest cover, bird species richness increased with reserve size irrespective of the level of urbanization in the surrounding matrix.

A number of relatively short-term studies (i.e., looking at data for periods of five years or less) have shown that forest cover requirements for forest-dwelling songbirds in fragmented landscapes varies depending on the species, and can range from 10 to 30%, with species persistence requiring at least 20 to 30% cover for most species (Andrén 1994; Fahrig 1997; both cited in Villard et al. 1999; Betts et al. 2007; Betts and Villard 2009; Tate 1998). However, current research based on longer-term data suggests that landscape forest cover requirements for forest birds range between 20 and 90%, with average requirements for long-term persistence being closer to 60% forest cover (Zuckerberg and Porter 2010). Other recent studies have determined that at least 40% forest cover is required to protect most sensitive forest bird species and sustain even some less sensitive species at healthy levels (Brown 2007; Rioux et al. 2009; Rompré et al. 2010). These data suggest that while there is significant species-specific variability, to provide habitat for most forest-dwelling birds in the context of eastern North America generally requires more than 30% forest cover.

Based on Cadman et al. (1987) and Riley and Mohr (1994), it is possible to compare the total number of species present to the number of species that could occur, based on their geographic ranges. At one end of the forest cover scale, 100% of the species that should occur were present in Ottawa-Carleton, which was approximately 30% forested. In contrast, Essex (then at 3% forest cover) had lost almost 40% of its forest birds. *The Ontario Breeding Bird Atlas* (*Atlas*) results (Cadman et al. 1987) were used to determine the number of forest-dependent bird species in municipalities with varying amounts of forest cover (an explanation of how to use the *Atlas* for local study areas follows in a subsequent section). Information related to the current (2005) atlas is available online at www.birdsontario.org/atlas/atlasmain.html. These data support the general concept that regions with lower levels of forest cover also support lower diversities of forest-dwelling birds.

The overall effect of a decrease in forest cover on birds in fragmented landscapes is that certain species disappear and many of the remaining ones become rare, or fail to reproduce, while species adapted to more open and successional habitats, as well as those that are more tolerant to human-induced disturbances in general are able to persist, and in some cases thrive. Species with specialized-habitat requirements are most likely to be affected adversely. In one study area near Ottawa, several species of forest birds disappeared as breeders when forest cover declined to below 30% (Freemark 1988). In Essex County, where there is about 3% forest cover, many wildlife species that are common to abundant elsewhere in Ontario are rare (e.g., Black-capped Chickadee and White-breasted Nuthatch [Oldham 1983]), and 80% of the forest-interior species have disappeared. In the Ottawa-Carleton area (with 30% forest cover), Hairy Woodpeckers can be found in woodlands 10 hectares or even smaller, whereas in the Town of Markham (with about 5% forest cover), none were found, even though some woodlands approached 100 hectares in area (Freemark 1988).

Current research has also begun to explore relationships between forest cover and amphibians. Studies on both mole salamanders and frogs have consistently found strong links between levels of local forest cover and both species diversity and abundance (e.g., Mazerolle et al. 2005), although these studies have typically been undertaken at a local scale (e.g., up to one kilometre, as shown in Table 10) rather than a landscape or regional scale. As with birds, variability among species has been observed. At scales of up to 1500 metres, researchers have found that populations were sustained at basic levels at about 30 to 40% forest cover, and that healthier levels of species diversity and abundance were correlated with levels of about 50 to 60% (Eigenbrod et al. 2008; Hermann et al. 2005; Veysey et al. 2009). However, for some amphibians, such as Wood Frog, the presence of that forest cover in close proximity to the breeding ponds appears to be as important as the level of cover (as shown in Table 10).

Table 10. Summary of Minimum Percent Forest Cover Requirements for Two Amphibian Species (from Homan et al. 2004)

Scale of Analysis (from breeding pool)	Spotted Salamander	Wood Frog
30 metres	32%	88%
100 metres	28%	78%
500 metres	41%	55%
1000 metres	51%	44%

The overall amount of forest cover in a landscape also helps determine its ability to support large mammals. Species such as Gray Wolf, Canada Lynx, and Elk that require extensive forests disappeared from southern Ontario shortly after forest clearing was initiated, and are now only found in central and northern Ontario where forest cover is more extensive. Some smaller mammals, such as Northern Flying squirrels, also require relatively high levels of forest cover in their home ranges (Ritchie et al. 2009), but landscape-scale data on many of these species are lacking.

In general, the literature indicates that a complex relationship exists between the relative importance of overall forest cover versus forest patch size and the ultimate response of individual wildlife species (e.g., Lee et al. 2002). However, on balance, the axiom is shifting away from “the bigger, the better” and towards “the greater amount of habitat within the landscape mosaic, the better” (see text box below, and Austen et al. 2001; Fahrig 2002; Golet 2001; Helms et al. 2009; Lindenmayer et al. 2002; Rosenburg et al. 1999; Vance et al. 2003; Veysey et al. 2009). These studies and reviews have all shown or suggested that forest patch size and shape play a lesser role in maintaining regional or landscape-scale levels of biodiversity than the total amount of forest cover, although the three metrics are to some extent interrelated.

Factors such as overall forest cover, forest patch area, shape and degree of fragmentation all affect the viability of habitat for wildlife species. However, for forest-dependent fauna, overall forest cover on the landscape is one of the most important habitat metrics. This means that in landscapes with relatively low levels of forest cover (e.g., less than 20%), some forest species will never return regardless of the spatial pattern of the remaining forest (Trzcinski et al. 1999).

Forest cover loss versus fragmentation

Forest habitat loss and habitat fragmentation are widely recognized as two key factors in the decline of wildlife species in the Mixedwood Plains and elsewhere. However, there continues to be uncertainty around the relative importance of habitat fragmentation in influencing thresholds in wildlife populations (Ewers and Didham 2006; McGarigal and Cushman 2002; Zuckerberg and Porter 2010).

One of the concepts that helps in understanding the relationship between forest cover, fragmentation and species loss is that of metapopulations. This is a term used to describe semi-isolated populations in a region that are linked by dispersion (Merriam 1988; Opdam 1991). Local extirpations of wildlife populations tend to occur naturally within forests due to failed reproductive efforts linked to (often stochastic) phenomena such as predation, parasitism, adverse weather conditions, natural catastrophes (e.g., fire and floods) and insufficient food. Where forest cover is fairly extensive and well-connected, forest patches naturally become recolonized by individuals from adjacent areas (so-called source-sink dynamics [Howe et al. 1991]). However, as the amount of overall natural area declines, recolonization can be constrained by lack of connectivity as well as reductions in the occurrence of colonizers, and extirpations may become permanent.

For breeding birds, it is increasingly evident that the amount of habitat in a given landscape is more important than the spatial configuration (i.e., extent of fragmentation) of that habitat in supporting the long-term persistence of forest birds (Betts et al. 2007; Donnelly and Marzluff 2004; Donnelly and Marzluff 2006; Fahrig 2002; Price et al. 2007; Radford et al. 2005; Zuckerberg and Porter 2010). This is supported by other studies that found forest patch area and edge effects did not significantly affect either nesting success or the productivity of neotropical songbirds (e.g., Friesen et al. 1998). Golet (2001) found that bird relative abundance was not predictable from swamp size, but found that the pattern of distribution was consistent with total forest availability. However, Lee et al. (2002) found that the relative importance of patch characteristics, patch area and landscape forest cover varied for different bird species, and others have found that once the level of overall habitat falls below a certain threshold, the level of fragmentation and spatial configuration of the remaining patches starts to take on more significance in supporting the remaining species in the landscape (Lichstein et al. 2002; Trzcinski et al. 1999).

For amphibians, metapopulation dynamics are important in regulating populations (e.g., Knutson et al. 2000), but the actual configuration of habitat in the landscape plays a more decisive role. Gibbs et al. (2005), in one of the few papers to examine long-term population trends at a landscape scale, found that Spring Peeper, Northern Leopard Frog and Wood Frog population persistence across New York State was associated with areas of increased deciduous forest cover. However, habitat configurations at relatively large scales (i.e., 5 to 10 kilometres) were associated with transitions in local populations, supporting the idea that metapopulation processes and fragmentation are also both important for the long-term viability of frog populations.

The metapopulations concept can also be used to explain the fact that the breeding bird assemblages in forests change annually (Villard et al. 1999). Common species are always present, but the more specialized species may be sporadic in occurrence. It has been demonstrated that—all else being equal—the number of breeding pairs in a region remains relatively constant, but that the areas used for breeding vary. Thus, a given forest patch may support a given species as infrequently as once every four or five years, yet this woodland is still critical to the overall maintenance of the regional populations. It follows that the disappearance of apparently insignificant forest patches may contribute to regional declines of forest bird populations.

2.3.2 Area of Largest Forest Patch

> Guideline

A watershed or other land unit should have at least one, and preferably several, 200-hectare forest patches (measured as forest area that is more than 100 metres from an edge).

> Rationale

The concept that large habitat reserves are superior to small reserves for the long-term persistence of area-sensitive forest species and low-mobility habitat specialists (e.g., Cottam et al. 2009; Keller and Yahner 2007; Price et al. 2007) continues to be supported. As well, it is increasingly recognized that high enough numbers of smaller patches in a landscape can also help support overall landscape biodiversity, at least for forest birds and amphibians, if the overall level of cover is above a certain threshold (as discussed in Section 2.3.1). Nonetheless, despite the new emphasis on the importance of overall forest cover, forest patch size remains a vital metric for forest species that are particularly area-sensitive or intolerant of human disturbances, or both.

Several studies have suggested that because the relative importance of patch area, patch characteristics and landscape cover varies for different species, these multiple factors should be considered in conservation planning (Andren 1996; Lee et al. 2002; Mörtberg 2001; Villard et al. 1999). For wildlife, small fragment areas (known as patches) impose a maximum limit on population size (and in birds, it would seem to be partially based on species reproductive rates [Vance et al. 2003]), which leaves species vulnerable to local extinction. Therefore, even though small patches can and do provide habitat for some species, preservation of some larger patches in the landscape is required for the long-term survival of forest populations as a whole. For example, some studies have identified only large (i.e., 500 hectares) or continuous forests as sources for Ovenbirds (Burke and Nol 2000; Mancke and Gavin 2000), even though they can breed successfully in smaller or "more fragmented" patches. Cottam et al. (2009) observed that in forest patches of more than 250 hectares, neither nest predation nor landscape matrix were significant factors in decreasing nesting success of Wood Thrush or Acadian Flycatcher, suggesting this area threshold was adequate for sustaining these species, at least in the short term.

Larger patches of forest also tend to have a greater diversity of habitat niches and area, and therefore are more likely to support a greater richness or diversity of both plant and wildlife species. For forest plants that do not disperse broadly or quickly, preservation of some relatively undisturbed large forest patches is needed to sustain them because of their restricted dispersal abilities and specialized habitat requirements, and to ensure continued seed or propagation sources for restored or regenerating areas nearby (Honnay et al. 2002; Jacquemyn et al. 2003; Taki et al. 2008).

Most studies looking specifically at forest patch area in relation to species diversity and abundance in the context of eastern North America have used birds as their focal species group. Robbins et al. (1989) determined that almost all forest birds in the mid-Atlantic

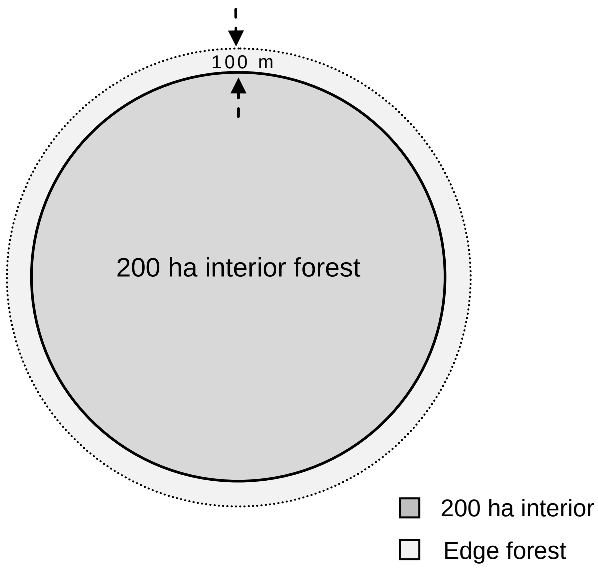

Figure 12. Two-hundred-hectare interior forest patch

states occurred at least occasionally in forests 100 hectares or smaller in area, but that the probability of detecting a number of these species in this size of patch was as low as 20 to 30%. Nol et al. (2005) examined the occurrence of 4 forest breeding birds (i.e., Ovenbird, Wood Thrush, Veery and Rose-breasted Grosbeak) in 216 woodlot fragments in southern Ontario. Neither vegetation features nor area of small woodlots adequately explained patterns of occupancy for any of the 4 species, and results suggested that maintenance of patches of at least 127 to approximately 300 hectares (i.e., with at least 90 to 230 hectares of "core" area—at least 100 metres from the forest edge) is needed to maintain source populations of forest breeding birds in the fragmented landscape of southern Ontario.

Weber et al. (2008) studied forest bird species occurrence in forest blocks of 3 size classes (i.e., less than 60 hectares, 60 to 100 hectares and more than 200 hectares) in a Maryland conservation network area. They found the majority (73%) of unlimited-radius plots with at least 5 species of forest birds were in forest blocks with at least 162 hectares and at least 120 hectares of "core" habitat.

In the Illinois Department of Conservation management guidelines for forest and grassland birds, Herkert et al. (2003) suggest that a 400-hectare forest patch is required to support 75 to 80% of the highly sensitive regional forest bird species, and predict that a 100-hectare forest patch would contain only about 60% of such species. Tate (1998) evaluated these guidelines by surveying 4 large forest patches ranging in size from 140 to 201 hectares in the Severn Sound Area of Concern. Tate found over 70% of the regional pool of forest bird species in the 4 forest patches collectively, and 79 to 87% of the expected forest-interior species in individual patches between 100 and 200 hectares in size. From this work, it was determined that a single patch of 100 hectares was too small to support the full complement of regional forest bird species, but that a forest patch of 200 hectares would be expected to support over 80% of all expected species in this region, and that maintenance of several tracts of forest of at least 200 hectares in the Severn Sound Area of Concern would support 90 to 100% of all expected species. Table 11 summarizes data from Tate (1998) and others relating patch size to forest bird habitat use.

In a unique study looking at forest patch size and bats, Henderson et al. (2008) found a positive relationship between total forest area, deciduous forest stands in particular, and the presence of Northern Long-eared Bats. They found that for every increase in 100 hectares of deciduous forest in a patch, the odds of a pair of bats occurring increased by 1.6 times, although no specific minimum area thresholds were suggested.

In general, at least for forest birds, the current science suggests maintenance of 100-hectare patches is inadequate, and that a moderate- to high-risk approach would be maintenance of at least a few 200-hectare patches in a given region or watershed, and at least one that is larger (i.e., closer to 500 hectares). Notably, this assumes that the patches are not long and thin, but also contain a good proportion of "core" habitat that is relatively undisturbed.

Table 11. Anticipated Response by Forest Birds to Area of Largest Forest Patch

Area of Largest Forest Patch*	Response by Forest Associated Breeding Birds
200 hectares	Will support approximately 80% of area-sensitive species including most area-sensitive species.
100 hectares	Will support approximately 60% of area-sensitive species including most area-sensitive species.
50–75 hectares	Will support some area-sensitive species, but several will be absent and species tolerant of edges will dominate.
20–50 hectares	May support a few area-sensitive species.
< 20 hectares	Few to no area-sensitive species. Dominated by species tolerant of edges.

* This assumes that these patches are circular or squarish in shape with some "core" habitat, not narrow linear patches.

2.3.3 Percent of Watershed that Is Forest Cover 100 Metres from Edge

> Guideline

The proportion of the watershed that is forest cover and 100 metres or further from the forest edge should be greater than 10%.

> Rationale

It continues to be generally agreed that the structure and functions of habitat edges are inherently different from those within habitat cores, and as a result, these areas tend to support a different number and range of species (Ewers and Didham 2006). However, there continues to be lack of certainty about where the "edge habitat" within a forest patch ends, and conflicting evidence about to what extent the presence of edge habitat in and of itself affects the long-term persistence of forest species in a fragmented landscape.

The main source of evidence for "edge effects" is from studies of forest-dwelling bird species. In a southern Ontario study, Sandilands and Hounsell (1994) determined that certain bird species avoided forest edges in small forests when they were breeding. In larger forests, one guild (or group) of species typically nested 100 metres or further from the edge, while a second guild consistently nested 200 metres or further from the edge. Subsequent work has supported these findings. For example, Austen et al. (2001) found that edge-intolerant species increased, and edge-tolerant species decreased, with both increasing woodlot size and core area, while Driscoll et al. (2005) found that for Wood Thrush increasing distance from forest-field edge was one of the four key metrics linked to nesting success.

"Edge effects" have been documented as extending from as little as a few metres into a forest patch, to more than several hundred metres, depending on the stressor, the effect being measured, the intervening habitat type and the sensitivity of the attribute (e.g., Batáry and Báldi 2004; Murcia 1995; Wood et al. 2006). However, in both scientific and technical studies, 100 metres from the forest edge is typically used as the generic measure of where measurable "edge effects" taper off, and where more undisturbed "core" habitat conditions begin (e.g., Dunford and Freemark 2004; Driscoll et al. 2005; Nol et al. 2005; Weber et al. 2008). While the 100 metres is not necessarily the most accurate measure, and fails to account for the effects of disturbances that may be internal to a given forest patch (e.g., as a result of human use of trails), it does provide a useful metric for generally assessing the overall extent of "core" habitat in a given landscape in relation to the extent of forested edge habitat.

Increased avian predation is also well-documented in forest edges, and is considered to be a reason some species have reduced breeding success in these zones. Studies have shown that brood parasites such as Brown-headed Cowbirds and avian predators such as snakes can be more abundant in forest edges (Batáry and Báldi 2004; Chalfoun et al. 2002a), and that depth or distance to the edge can affect forest breeding bird diversity and abundance (Mancke and Gavin 2000).

Some studies report conflicting responses of forest bird species to forest edge habitats. For example, Chapa-Vargas and Robinson (2007) reported that Acadian Flycatchers seem to prefer edge habitats, and show no negative effects with reduction in presence or extent of cores, while Hoover et al. (2006) observed "edge effects" (i.e., Brown-headed Cowbird nest parasitism extending 600 metres into the forest edge, but most pronounced at 250 metres from the edge) for this same species. Similarly, Kaiser and Lindell (2007) and Friesen et al. (1999) found no effect of edge type or distance to edge on nest survival for Wood Thrush.

Some studies have found evidence that edge avoidance is linked to overall density of the species within the patch (Bollinger and Switzer 2002). Although the literature is relatively consistent on the increased negative effects of Brown-headed Cowbird nest parasitism and avian predators on edge-nesting birds, Chalfoun et al. (2002b) found that there were no differences in small- and medium-sized mammalian predator abundance between edge and core forest habitat. A recent study cautioned that the abundance of a potential nest predator species does not automatically equate to actual risk of predation for a songbird species, and that the list of species regularly identified as common nest predators in urban and fragmented locales may require revision with respect to their individual importance (L. Friesen, Canadian Wildlife Service, Burlington, pers. comm. 2012).

Perhaps the explanation for the variation in the findings of these studies lies more in the nature of the matrix. Ewers and Didham (2006) conclude from their review that the more the matrix differs from the habitat patch, the more that edge effects will be evident. Conflicting study results may also be explained by the theory that, at the community level, birds at more productive sites differentiate across a gradient in edge, whereas bird communities at less productive sites do not (McWethy et al. 2009). They may also be explained by erroneous assumptions about expected responses of certain species to different habitat scenarios (e.g., perhaps Wood Thrush and Acadian Flycatcher are more generalist than previously thought), or by lack of attention to the matrix (i.e., current findings suggest that "edge effects" decrease as overall forest cover increases, as discussed in Section 2.3.1).

Table 12 indicates how forest bird species can be affected by differing percentages of non-forest cover. Notably, when forest cover declines to around 15% (in combination with fragmentation into smaller forest patches), 20 to 25% of area-sensitive species disappear. An exception is Haldimand and Norfolk, which continues to support a high percentage of forest-breeding birds even with relatively low levels of overall forest cover. However, this can be partly explained by the fact that these counties contain several large (i.e., more than 1000 hectares) forests in relatively close proximity, and several areas within the county contain over 30% forest cover, providing further support for the patch area guideline (in Section 2.3.2).

Table 12. Number of Forest-Associated Bird Species in Five Areas of Southern Ontario with Differing Percentages of Forest Cover (Adjusted for Potential Breeding Ranges Based on Pre-settlement Habitat)

	Ottawa-Carleton	Haldimand and Norfolk	Waterloo and Wellington	Middlesex	Essex
Percent forest cover	29.4	16.2	14.8/18.2	13.5	3.0
Total number of species within range	94.0	102.0	100.0	102.0	102.0
Number of species occurring	94.0	98.0	88.0	83.0	63.0
Percent of total number of species within range present	100.0	96.1	88.0	81.5	61.7
Number of FIE and FI species within range	60.0	66.0	64.0	61.0	66.0
Number of FIE and FI species present	60.0	62.0	54.0	50.0	36.0
Percent of FIE and FI species within range present	100.0	93.9	84.4	82.0	54.5
Number of FI species within range	18.0	20.0	20.0	20.0	20.0
Total FI species present	18.0	19.0	15.0	16.0	4.0
Percent of FI species within range present	100.0	95.0	75.0	80.0	20.0

FIE - Forest interior/edge
FI - Forest interior
Source: Cadman et al. (1987); Riley and Mohr (1994)

Irrespective of the gaps in our understanding of this issue, the presence of an adequate level of "core" habitat is still considered a worthwhile metric to use in landscape-scale natural heritage planning (e.g., Nol et al. 2005; Tate 1998; Weber et al. 2008). This is largely because forested areas with large "core" areas (and less "edge") also, by default, tend to be more diverse habitats capable of supporting a broader range of habitats and species, and also tend to be less disturbed and therefore able to support species that are sensitive to human-induced disturbances (Imbeau et al. 2003). Such areas can accommodate a range of habitats and micro-habitats created by stochastic events (e.g., tree falls), as well as a range of successional stages of each habitat type, each with the potential to support different species. The presence of these elements in a landscape or watershed enhances biodiversity, and increases resilience to natural stressors such as diseases or insect infestations.

2.3.4 Forest Shape

> Guideline

To be of maximum use to species such as forest-breeding birds that are intolerant of edge habitat, forest patches should be circular or square in shape.

> Rationale

Although forest shape, as a stand-alone metric, has been poorly studied (Ewers and Didham 2006), it is generally accepted that shape is directly related to the concept of "edge effects" and the extent of contiguous core habitat, as discussed in the previous sections. Figure 13, on forest shape determining amount of core habitat, illustrates how habitat shape can influence the amount of core habitat; square or circular habitats provide the greatest amounts of core habitat compared to the area of habitat that is influenced by edge, while similarly sized linear or irregularly shaped habitats may contain little or no core habitat.

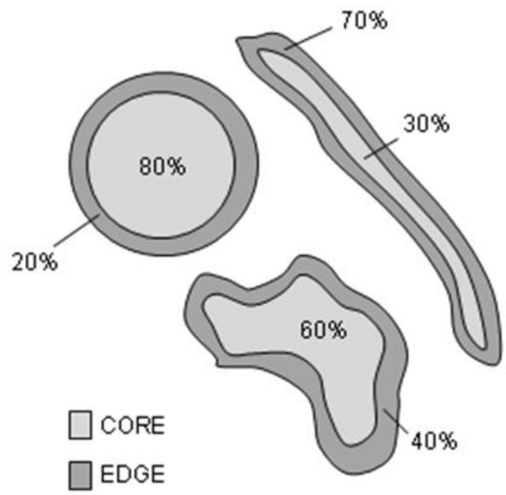

Figure 13. Forest shape determines the amount of core habitat – avoid narrow linear shapes for maximum core habitat (Kennedy et al. 2003)

Linear or irregularly shaped forest patches in a fragmented landscape tend to be colonized more frequently and more extensively by predators (particularly in urban environments), more vulnerable to invasive species, and subject to lower rates of reproduction and more wildlife species emigration (Batáry and Báldi 2004; Bowles 1999; Deng and Gao 2005; Hylander et al. 2002; Weldon 2006). This ultimately results in reduced population persistence of indigenous species adapted to forest habitats.

This is not to say that linear or irregularly shaped forest patches do not provide habitat for a diversity of species. In fact, forest patches dominated by edge and successional type habitats have been shown to provide excellent habitat for species that are less specialized in their requirements, as well as those that thrive in successional areas (Angold et al. 2006; Carafagno and Weatherhead 2006; Chapa-Vargas and Robinson 2007; Imbeau et al. 2003; Rioux et al. 2009). However, a predominance of these types of forest patches excludes those species and guilds that require more extensive core forest habitat area for successful foraging and reproduction.

From this perspective, it is clear that in terms of restoration opportunities, the "infilling" of irregular forest patches can offer some potential wildlife benefits in terms of increasing the extent of core habitat (and decreasing the extent of edge habitat).

2.3.5 Proximity to Other Forested Patches

> Guideline

To be of maximum use to species such as forest birds and other wildlife that require large areas of forest habitat, forest patches should be within two kilometres of one another or other supporting habitat features.

"Big Woods" areas, representing concentrations of smaller forest patches as well as larger forest patches, should be a cornerstone of protection and enhancement within each watershed or land unit.

> Rationale

This guideline addresses four metrics:
(1) minimum distances between patches,
(2) distance to the nearest productive patch,
(3) differing mobility of different species or taxonomic groups, and (4) the matrix as a barrier (e.g., busy roads can present a significant barrier between habitats for most forest-dwelling herpetofauna, but less so for most forest birds).

The underlying hypothesis beneath this guideline is that one of the key effects of habitat fragmentation is that patches of habitat become isolated from one another, particularly as overall levels of cover decrease, disrupting natural species distribution and migration patterns, and requiring dispersing individuals or seeds, spores, bulbs or roots to traverse an intervening matrix. Habitat isolation in both space and time disrupts species distribution patterns, with consequent effects on metapopulation dynamics and the genetic structure of patch-dwelling populations (Ewers and Didham 2006).

For forest birds, this distance between patches is related to the theory of source and sink populations. This is a relatively long-standing theory that has recently been re-examined for forest birds in fragmented landscape in southern Ontario (e.g., Nol et al. 2005). The theory is that for some species of forest-breeding birds, productivity is much higher in large forest patches than in small forest patches, and is higher than the level needed to replace individuals within the larger patches. These then serve as "sources" for populating neighbouring, smaller patches ("sinks") where productivity is lower and not adequate for replacement.

Most forest plants have restricted dispersal abilities and require specific conditions for seedling establishment (Honnay et al. 2002; Jacquemyn et al. 2003), which makes them vulnerable to isolation in the context of fragmentation. Endels et al. (2007) looked at understorey species in a Belgian forest with a long history of disturbances and found that spring-flowering herbs (with large seeds and unassisted dispersal) were effectively isolated by their limited ability to colonize young forests.

For forest birds, research has found that habitats in close proximity to other natural areas support more species than isolated habitats of the same size, and that some species with large home ranges may use several patches instead of one large area. Recolonization of habitat patches by

Scarlet Tanagers (a forest-interior species) was found to decrease as the isolation of patches increased (Hames et al. 2001). Austen and Bradstreet (2006) found abundant forest cover within two kilometres of a particular forest patch was a significant predictor for the presence of bird species that prefer interior forest habitat in Norfolk County. Nol et al. (2005) found that for Ovenbird, Wood Thrush, Veery and Rose-breasted Grosbeak, with each kilometre increase in distance from a large patch, the probability of occurrence in small patches decreased, and that small occupied patches were almost all within 14 kilometres, and on average were within around 5 kilometres, of each other. Driscoll et al. (2005) found that one of the four key variables influencing Wood Thrush nesting success in a rural area of New York (with just over 50% forest cover) was total core habitat located within 5 kilometres.

Forest amphibians might be expected to be more vulnerable to habitat isolation than birds in the context of fragmentation because of their relatively more restricted mobility. However, the few relevant examples of current research conducted at large scales shows that they have a surprising capacity to move relatively long distances, even in fragmented landscapes. Eastern Newts were found to require suitable habitat to complete their life cycle needs within 5000 metres of their original breeding pools (Rinehart et al. 2009), while Hermann et al. (2005) reported that pond breeding amphibians require forested habitat within 1000 metres of their breeding pools to ensure healthy local levels of diversity and abundance. In a unique study examining frog population trends over a vast area and a period of 30 years, Gibbs et al. (2005) found that frog metapopulation processes occur at much larger scales than expected (i.e., up to 10 kilometres).

In landscapes with relatively low forest cover overall, species diversity and survivorship increase where the remaining habitat patches are larger and more clumped or aggregated. In the eucalyptus forests of Australia, Radford et al. (2005) found that landscapes with fewer larger patches in a more clumped distribution tended to have a less pronounced extinction threshold occurring at a higher proportion of habitat loss. Donnelly and Marzluff (2006) reported that in the urban setting of Seattle, Washington, bird species richness was high and many native forest plant species were retained where landscape-level forest cover was at least 48% (including trees outside natural areas) and the remaining forest was at least 64% aggregated across the landscape. In their examination of breeding bird data across eastern North America over a 20-year period using 25 kilometres square scale analyses, Zuckerberg and Porter (2010) found that of the 22 forest bird species demonstrating persistence thresholds, 16 of them exhibited strong support for the inclusion of the "clumpiness" variable in the threshold model, indicating a preference for areas where forested habitats were concentrated.

Based on the limited available science, the isolation between forest patches for forest birds generally occurs at about five kilometres, but for amphibians at between one and two kilometres. For forest plants, some level of immediate proximity is required, since effective colonization of many spring ephemerals will not effectively occur under conditions of fragmentation, and so patches must be connected by wooded corridors (as described in Section 2.3.6). For landscapes with relatively low forest cover, species diversity and survivorship improves where the remaining habitat patches are both more extensive and more clumped or aggregated.

Big Woods areas

The literature speaks strongly to the importance of concentrations of forest patches and the inclusion of larger habitat patches in landscapes to serve as sources of, as well as reserves for, forest species with specialized or area-sensitive requirements. The concept of "Big Woods" (Mancke and Gavin 2000) recognizes and elevates the importance of aggregations of forest patches, and can provide guidance for future replacement efforts. Two related metrics form the basis of a Big Woods approach: the aggregation of forest patches and the identification of the largest forest patches. Additional metrics for assessing forest function may include the measurement "distance to nearest Big Woods" and level of disturbance within the "Big Woods."

A variation on this method has been used in the Lake Simcoe Basin Natural Heritage System (Beacon Environmental and LSRCA 2007) and the East Gwillimbury Natural Heritage System where a Big Woods Policy Area was established in the Official Plan to help emphasize the importance of smaller forest patches within high forest-cover areas and provide focus areas for restoration activities.

2.3.6 Fragmented Landscapes and the Role of Corridors

> Guideline

Connectivity width will vary depending on the objectives of the project and the attributes of the forest nodes that will be connected. Corridors designed to facilitate species movement should be a minimum of 50 to 100 metres in width. Corridors designed to accommodate breeding habitat for specialist species need to meet the habitat requirements of those target species and account for the effects of the intervening lands (the matrix).

> Rationale

Dispersal between forest habitat patches is recognized as essential for long-term metapopulation persistence of species in fragmented landscapes (Noss and Harris 1986; Riley and Mohr 1994). The basic premise that an effective conservation strategy connects existing high-quality nodes using actual and potential corridors still persists (Townsend and Levey 2005). Determining the nature of those connections, based on species behaviour and life history, remains a challenge. Current thinking includes the "circuit theory" hypothesis, which suggests that for most species there are a multitude of potential connective pathways (McRae and Beier 2007) and that certain parts of a landscape offer different degrees of resistance and opportunity for species movement. An animal using a corridor likely has no route envisioned as humans would, but moves onward instinctually and is faced with multiple sequential decisions as to the best immediate pathway based on the "least cost" to itself.

The utility of corridors to facilitate movement of forest species between forested areas in a fragmented landscape continues to be debated (e.g., Angold et al. 2006; Beier and Brost 2010; Hannon and Schmiegelow 2002; Whitfield 2001). The emerging evidence shows that although corridors may be used by many species to facilitate movement, they are not required by all species or in all landscape contexts (Davies and Pullin 2007; Falcy and Estades 2007). It also shows that the use of designated corridors may depend on the permeability of the matrix (i.e., corridor use in urban landscapes may be more evident and important than in rural landscapes where species may move across open or remnant successional habitats) (Gilbert-Norton et al. 2009; Rizkalla and Swihart 2007).

Some species are obligate users of corridors, either being totally dependent upon them to get from one natural patch to another or highly preferring to use them to get across the landscape. Habitat connectivity and corridors in fragmented landscapes have been shown to benefit a number of groups of species (e.g., some plants, amphibians and reptiles) (Damschen et al. 2006; Damschen et al. 2008; Haddad et al. 2003; Milne and Bennett 2007; Veysey et al. 2009). Gilbert-Norton et al. (2009) found, in their meta-analysis of 35 corridor studies, that corridors can increase movement between patches by approximately 50% compared to unconnected patches. They also found corridors were more important for movement of invertebrates, non-avian vertebrates and plants than they were for birds.

Corridors can also facilitate movement for some undesirable species (e.g., invasive plants and pathogens), may increase exposure to predators (e.g., Weldon 2006) and may be of limited use to birds, some forest plants and some mammals. In a context of regional forest cover of close to 40%, with wooded patches of 3 to 165 hectares that were no more than 325 metres apart, Scarlet Tanagers in a rural landscape did not utilize corridors to facilitate movement between patches (Fraser and Stutchbury 2004). Honnay et al. (2002) found almost all forest plant species (85%) had extremely low success in colonizing solitary new suitable forest habitats even after 40 years. In a landscape with higher forest connectivity, colonization success was higher but still insufficient to ensure large-scale colonization. Some species such as Red Fox and Coyote often move through open habitat, while White-tailed Deer tend to travel through open habitat or along a corridor or corridor edge if it happens to be leading in a direction that offers food, shelter or other benefits.

The determination of optimum corridor widths for wildlife movement is challenging, and the literature provides limited guidance with respect to appropriate minimum widths, lengths or width-to-length ratios. Wooded corridors 50 metres in width can facilitate movement for common generalist species. In terms of stream corridors, widths of 75 to 175 metres have been suggested for breeding bird species, and 10 to 30 metres were found to be sufficient to include habitat for 90% of streamside plant species (Spackman and Hughes 1995). Mason et al. (2007) examined species occurrence in long, narrow and largely forested greenway corridors in an urban context in North Carolina and found that urban-adapted species and edge-dwelling birds (i.e., Mourning Dove, House Wren, House Finch and European Starling) were most common in greenways less than 100 metres wide, while forest species (e.g., Acadian Flycatcher, Hairy Woodpecker and Wood Thrush) were not recorded in greenways narrower than 50 metres, and were primarily in greenways wider than 100 metres. Other forest species (e.g., including ground nesters such as Black-and-white Warbler, Louisiana Waterthrush and Ovenbird) were only recorded in greenways wider than 300 metres. The need to look at individual species needs and the range of factors that need to be considered is illustrated by Veysey et al. (2009), who found that Spotted Salamanders were able to enter and to successfully cross clear cuts between forested patches of up to 100 metres wide, but only under conditions of adequate precipitation.

Width and length determinations are further complicated by the difference between the intrinsic habitat values that may be found within linear habitat patches (e.g., breeding habitat for area-sensitive breeding birds), and the narrower function of movement (e.g., by plants through pollination and seed dispersal, or animals) along a pathway that facilitates movement from one node to another. An area of only one metre in width may be used as a travel corridor by some wildlife species, while other species that must breed within corridors (e.g., some salamanders, small mammals or insects) may require much wider features that will support productive breeding habitat.

Corridor widths should ultimately be determined based on a functional assessment of what the corridor is expected to achieve, and what species it is expected to accommodate. Considering only movement, a minimum guideline of 50 to 100 metres is supportable. The provision of breeding habitat for target species would require knowledge of patch size requirement and an analysis of the potential for edge effects. In rural landscapes, it has been suggested that corridors should be as wide as 500 metres for specialist species, although this approach begins to overlap corridor function with other functions such as habitat patch size and shape. Intuitively, in urban environments, wider corridors would be required to provide the same level of function in the face of urban effects.

Corridors for wildlife must also provide suitable habitat for the species that are expected to move along them. Ideally, vegetation composition in the corridor should be similar to that in the corridor nodes (or reflect soil/historic conditions), should be continuous between nodes and should be a minimum width along its entire length. The current literature also points to the higher effectiveness (i.e., levels of use) of natural or existing features versus purpose-created corridors, which supports the preservation and integration of remaining habitat fragments (wooded riparian areas, use of "stepping stone" woodlots, hedgerows, etc.) over the creation of new corridors, particularly to facilitate movement of forest-dwelling species (Gilbert-Norton et al. 2009).

In general, the development of a corridor strategy should consider landscape features and attributes (such as natural cover and the composition of surrounding matrix), the different dispersal abilities and mobility of the different species being accommodated, corridor opportunities and constraints, as well as both positive and negative potential ecological outcomes from corridor creation.

2.3.7 Forest Quality: Species Composition and Age Structure

> Guideline

Watershed forest cover should be representative of the full diversity of naturally occurring forest communities found within the ecoregion. This should include components of mature and old-growth forest.

> Rationale

Using remote sensing and GIS, quantitative measures such as percent forest cover or proportion of forest cover within 100 metres from a patch edge can be readily measured. Collecting qualitative information such as species composition, structure and age structure of a forest requires ground-truthing and site-specific data collection, which is generally more costly and time consuming than remote sensing analyses. However, this information can be just as important, or more so, in terms of fully assessing the ability of a given watershed or region to support various species. For example, forest cover may be plentiful in a particular watershed, but may be dominated by early to mid-successional plant communities, conifer plantations and a variety of non-native species, and will therefore be limited in terms of the range of biodiversity that it can support.

Cushman et al. (2008) found that forest cover types in western Oregon determined from satellite imagery were a weak proxy for habitat, explaining only 4% of the variance in bird species abundance. Therefore, it is important to collect and integrate any additional information about the nature of the forest cover into natural heritage planning wherever possible.

The literature speaks to the importance of forest composition and the presence of native plants in relation to plant, insect and bird species diversity in both rural and urban landscape contexts. In urban settings, Angold et al. (2006) found that linear greenways with more native species and more structural diversity were used by a greater diversity of insects, while Donnelly and Marzluff (2004; 2006) found positive correlations between proportions of native vegetation species and abundance of native bird species. Austen and Bradstreet (1996) found differences in forest composition, as defined by proportion of deciduous-to-coniferous and swamp-to-upland forest, were important for individual bird species. For example, Veery and American Redstart were found in patches with more deciduous cover, while Blackburnian Warbler, Pine Warbler and Ovenbird were found more often in woodlands with more coniferous forest.

The literature also speaks to the importance of mature forests for certain species, and the importance of a diversified forest structure. Betts and Villard (2009) found a strong association between five forest warblers and the presence of mature forest, while Ritchie et al. (2009) observed flying squirrels more frequently in mature than in regenerating forest patches, and Baldwin et al. (2006a) found Spotted Salamander richness was associated with more mature forests having more forest litter and organics. Jaquemyn et al. (2003) reported 50 out of 59 investigated forest plant species had a significantly higher occurrence in older forest patches than regenerating forest patches, indicating that there is a time lag between patch regeneration and effective colonization of restored patches for most forest plants. Soil acidity, nutrient and moisture levels were all important factors. The presence of a shrub layer has also been repeatedly linked to greater levels of forest bird diversity and nesting success (Donnelly and Marzluff 2004; Driscoll et al. 2005; McWethy et al. 2009).

Weber et al. (2008) conducted a study of forest bird occurrences in variously sized forest blocks in a conservation network in Maryland and found that forest bird richness was higher at sites where: disturbance was minimal (i.e., no more than 5% of the area); deciduous trees were dominant; seasonally, semi-permanently or permanently flooded wetlands were present; perennial water, including streams, ponds or permanently flooded wetlands, were present; and stands were relatively mature (i.e., mean canopy diameter at breast height was at least 40.6 centimetres and tree saplings were less abundant). Forest bird richness was also positively correlated with canopy tree richness, and the length of streams and the area of wetlands within one kilometre.

These studies show fairly consistent support for higher species diversity, and particularly for the relatively well-studied bird diversity, in forest patches that are more mature, have more native species, and that contain or are close to permanent or intermittent water features. Milne and Bennett (2007) also found that more contiguous, native deciduous forests supported higher numbers of forest-breeding birds than other wooded areas in a southern Ontario watershed, but noted that coniferous plantations also provided breeding habitat for a number of bird species (e.g., Golden-crowned Kinglet, Pine Warbler, Yellow-rumped Warbler).

Site conditions, such as soil and topography, should also play important roles in determining which habitat types to restore in a particular area. In order to guide forest and wetland restoration in the Niagara River Area of Concern, Environment Canada used soil drainage categories to determine the original proportion of upland to lowland forest present (Snell et al. 1998). Due to drastic losses of upland forest, they recommended that restoration focus on drier vegetation communities. Deciding which forest types are priorities for restoration requires a sense of the pre-settlement landscape as guidance, in the same manner in which a cumulative impact analysis was recommended for wetlands prior to decisions being made on wetland restoration projects (Bedford 1999).

2.4 Grassland Habitat Guidelines

Grassland is defined here as areas with less than 35% tree and shrub cover and includes native prairie, savannah, old fields and meadows, pasture and hay. Alvars are a related habitat and meadow-type open alvars share many grassland characteristics, scientific attention and conservation concerns, and for the purposes of this report are considered grassland habitat. The grassland characteristics of shrub and treed alvars need to be assessed locally at an individual site scale. Of all the so-called open country habitats, only shrublands with more than 35% cover by shrubs, orchards, row crops and cut-overs are excluded from this discussion.

Southern Ontario has always been a changing environment, from glaciation to a forest biome interspersed with grassland patches and landscapes to the matrix of agricultural lands that we see in many areas today. More recently the extent to which First Nations manipulated the forest biome for agricultural purposes has come into question. There is some evidence that large areas of the landscape were cleared to grow corn, among other uses. In one example, the estimated extent of agricultural activity associated with a cluster of First Nations settlement near Stouffville, Ontario, was thought to have extended over 40 square kilometres (R. Williamson, Archaelogical Services Inc., Toronto, pers. comm. 2012). While there is uncertainty as to what extent grasslands formed the dominant local land cover, it is likely that large areas of present-day southern Ontario pre-European contact were suitable habitat for grassland birds.

An estimated 100 000 hectares of tallgrass prairie was once found in southern Ontario, while today less than 3 000 hectares remain (Tallgrass Ontario, www.tallgrassontario.org). This likely represents less than 0.15% of grassland habitat that is present today. For example, in 2011, approximately 1.5 million hectares in Ontario were hay, pasture or fallow according to the Ontario Ministry of Agriculture, Food and Rural Affairs (www.omafra.gov.on.ca/english/stats/agriculture_summary.htm). If non-agricultural grasslands are considered, it is likely in 2011 that over 2 million hectares of grassland habitats (as defined here) were extant in southern Ontario alone. Clearly, even compared to the original extent of tallgrass prairie, the agricultural habitats (often referred to as "surrogate" habitats) are the most important driver for grassland breeding birds today and will likely be into the future.

Grassland as a unique land use and land cover

While the exact land cover composition of pre-European Ontario is unknown, today there is no doubt that open country is the predominant landscape in many areas of the Mixedwood Plains. Much of the present open country is overwhelmingly non-native and largely agriculturally derived (Neave and Baldwin 2011). Many grassland species, notably birds, do use this abundant non-native land cover as habitat while other species, notably plants and insects, prefer, or are restricted to, remnant prairies, savannahs and alvars.

For species that thrive in the agricultural landscape, their future presence and populations will be determined by their adaptation to changing land use and land cover. Nearly all grassland patches have simultaneous wildlife and human uses, and land cover is heavily linked to land use. Other habitats in the Mixedwood Plains do not necessarily have the same level of concurrent human use.

For the most part, grassland habitat is maintained through active disturbance, including agricultural land use. Unlike forests, grassland can be ephemeral, being created and lost within the same year or over a period of weeks, and the location of patches may vary from season to season. Under this disturbance regime, a temporal detachment between land use and land cover can be seen. For example, a field may be in hay production all season but the actual land cover of tall grass and its associated habitat value may change suddenly throughout the season depending on hay harvest times and frequency.

The relative area devoted to different agricultural crops and farm practices is determined by economic and social forces. These forces act at local, provincial, national and international scales. Both urban and rural Ontario underwent significant change over the last century. Throughout the 20th century, expansion and contraction of agriculture have occurred. Forest and shrubland have reclaimed most abandoned farmland. The relative area devoted to row crops, such as corn, wheat and soybeans, versus forage crops, such as hay and pasture, will change with economic demand. The numbers of farms and the human population that is actively engaged in farming have declined with growing efficiency and specialization, while the non-farm rural population has grown significantly. All of these factors influence the nature and extent of grasslands.

Land use planning can be used to restrict activities in order to maintain natural land cover such as forest and wetland, but it has not traditionally been used to require activities such as grazing or set-asides, to create or maintain a land cover or wildlife habitat. As agricultural grassland habitat is a created and managed land cover, there is little regulatory or planning onus to continue to maintain or create this cover, although there are many examples of programs aimed at precisely this, either through policy or incentive, both in North America (e.g., the Canadian Permanent Cover Program) and elsewhere (e.g., Europe).

Introducing the grassland guidelines

Addressing grassland habitat in the *How Much Habitat is Enough?* series has not been previously contemplated. Notwithstanding the above-noted realities, in this edition, the subject is being broached by concentrating on native grasslands in the knowledge that, as our understanding of the role of the agricultural landscape evolves, the guidelines will need to be further expanded. The guidance is therefore targeted at public or private lands with conservation potential. While this foundation is meant to target the lost diversity of imperilled native grasslands and the species that are dependent upon them, it is anticipated that they will complement and assist in the larger discussion regarding the overall quantity of non-native grasslands.

By necessity, in eastern North America, most of the literature has been drawn from the agricultural landscape and the birds that occupy that landscape. In particular, these guidelines address habitat type, amount and extent, as well as landscape configuration and patch size.

Table 13. Summary of Grassland Guidelines

Parameter	Guideline
Where to protect and restore	Focus on restoring and creating grassland habitat in existing and potential grassland landscapes.
Habitat type and area	Maintain, restore and create native grassland patches to their historic extent and type at a county, municipal and/or watershed scale considering past presence and current conditions.
Landscape configuration, heterogeneity and connectivity	Grassland habitat patches should be clustered or aggregated, and any intervening land cover should be open or semi-open in order to be permeable to species movement.
Patch size	Maintain and create small and large grassland patches in existing and potential local grassland landscapes, with an average grassland patch area of greater than or equal to 50 hectares and at least one 100-hectare patch.
Landscape heterogeneity	Some grassland habitat should be located adjacent to hedgerows, riparian and wetland habitats for species that require different habitat types in close proximity.

Open country

How Much Habitat is Enough? addresses grassland habitat that is part of a land cover and land use category known as open country. Open country is the dominant landscape matrix of the Mixedwood Plains and is comprised of any vegetation community where the combined cover of trees and shrubs over 1 m tall is less than 60%. Open country can be expressed as land use, e.g., a hay field, or as a land cover, e.g., grass or hay. Within this category is a variety of land uses and land covers including pastures, hay fields, old fields, prairies and savannas, alvars, rock barrens, burned areas, grains, thickets, and shrublands. Not all open country has equal value as habitat: e.g., some row crops such as corn have very limited habitat value for breeding birds, but migrating or wintering waterfowl will use post-harvest waste grains as food.

2.4.1 Where to Protect and Restore

> Guideline

Focus on restoring and creating grassland habitat in existing and potential grassland landscapes.

> Rationale

Of all the broad types of native habitats and ecosystems represented in the Mixedwood Plains, there are none that have been so diminished in extent as native grassland communities. Ninety-seven percent (97%) of the original native grassland habitat in southern Ontario has been lost (Bakowsky and Riley 1994). In this guideline, the restoration and creation of grassland patches first focuses on existing and potential local grassland landscapes: those where native grassland communities were historically, or are, the dominant local vegetation feature and/or have a high potential to be re-established.

Existing and potential local grassland landscapes:

- Have existing or historic native grassland that was a major or dominant local habitat feature;
- Are areas with existing populations of grassland species, particularly species at risk; and
- Have favourable current conditions and land uses to maintain, manage or restore large areas of grassland.

These are the target areas to apply the initial guidance in this report.

Historically, landscapes such as the Carden Plain often had large areas of native grassland, and today have large native and agricultural grassland patches. These areas are generally not prime agricultural lands and/or support activities such as grazing or forage crops with current or potential value as suitable agricultural grassland habitat. In these areas, there may be greater opportunities for grassland restoration and maintenance. Likewise, in these grassland landscapes, forest-based restoration targets may be less locally applicable, may already be met, or at least need to be balanced with grassland considerations.

2.4.2 Habitat Type and Area

> Guideline

Maintain, restore and create native grassland patches to their historic extent and type at a county, municipal and/or watershed scale considering past presence and current conditions.

> Rationale

Native grassland is regionally rare but a part of the larger ecosystem, responding to a particular set of environmental conditions. It adds to the scope and diversity of the ecozone, hosting unique species. While some birds will use both native and agricultural habitats, other groups have less flexibility. For example, many invertebrates have particular host plant requirements that are only met in native habitats, such as Karner Blue butterfly with Wild Lupine, and Giant Swallowtail butterfly with Common Hoptree. In addition, other species rely on structural and species diversity only found within native grassland habitat patches. These patches are of value whether they are part of local grassland landscapes, or as naturally occurring meadows or other openings in forested landscapes.

Historically, native grasslands were not extensive in the Mixedwood Plains, with only an estimated 100 000 hectares of prairie in southern Ontario (Neave and Baldwin 2011) as well as some meadow- and grass-dominated alvars. Today many grassland species of plants and insects and some birds are rare or endangered. The minimum amount of native grassland required for these species to persist is largely unknown. However, it is likely that many species of conservation concern that rely on native grassland habitat in Ontario are already below any theoretical population and habitat thresholds for long-term persistence. Any gain in the amount of native grassland will be expected to have positive effects on populations of species other than birds.

Beyond local landscapes and native grasslands

In terms of birds, landscapes with a greater area of native or non-native grasslands contain greater avian diversity than landscapes dominated by other agricultural covers such as row crops (Best et al. 2000; Lindsay et al. in review). Studies of the changes in bird populations since the initiation of the Conservation Reserve Program (CRP) in the United States and the Permanent Cover Program (PCP) in Canada show the value of grassland to the landscape. Birds have higher abundances, densities and reproductive success in these restored grasslands than in the croplands they replaced (Best et al. 1997; Johnson and Igl 1995; King and Savidge 1995; Patterson and Best 1996). For example, in Illinois, the creation of undisturbed grassland habitat by the CRP resulted in a 10-fold increase in the Henslow's Sparrow population and a 5-fold increase in the Grasshopper Sparrow population (Herkert 1997; Herkert 1998; Herkert et al. 2003). In Minnesota, Haroldson et al. (2006) found that for each 10% increase in landscape grass cover, the counts of meadowlarks in the summer increased on average by 11.7 birds per route. For 5 duck species, Reynolds et al. (2001) found higher nesting success and recruitment in CRP lands. In the Canadian prairies, McMaster and Davis (2001) found that 9 of the

10 commonly occurring grassland birds were more abundant on PCP sites than cropland sites, and PCP also had higher bird species richness than cropland. The CRP- and PCP-related studies show the habitat value of grassland.

In general, the probability of species occurrence, species abundance or nest productivity increased with the amount of grassland or less intensive agriculture on the landscape (Table 14). Tews (2008a) found that 6.5% suitable habitat within a 100-kilometre-square landscape was required to maintain a Bobolink population with less than a 5% probability of extinction over 50 years, and that this amount increased to 30% suitable habitat within a 100-kilometre-square landscape to maintain a functional population.

Table 14. Species Presence, Abundance or Population Parameters and the Amount of Habitat in the Landscape

Species	Relationship with Amount of Habitat in the Landscape
Northern Harrier	Abundance positively correlated with pasture and hayfield (2); Positive with grassland amount at 800 m scale (1)
Savannah Sparrow	Abundance positively correlated with amount of natural and seeded pasture and decreased when crop area greater than 60 to 70% in agricultural lands (abundance increased from 0.53/ha in lands with 0.1 to 3% pasture to 0.69/ha in lands with 8 to 28% pasture and from 0.35/ha in lands with 1 to 20% crop to 0.87/ha in lands with 53 to 84% crop (3); Occurrence negatively correlated with percent trees and shrubs in landscape (4); Occurrence positively correlated with grassland cover in 32 km^2 landscape (5)
Eastern Meadowlark	Abundance positively correlated with amount of natural and seeded pasture and decreased when crop area greater than 40% in agricultural lands; Relative abundance increased from 0.11/ha in lands with 0.1 to 3% pasture to 0.39/ha in lands with 8 to 28% pasture (3)
Western Meadowlark	Occurrence positively correlated with grassland and hay cover within 8 km^2 (5); Abundance positively correlated with grassland cover (6)
Bobolink	Abundance positively correlated with amount of natural and seeded pasture and decreased when crop area greater than 40% in agricultural lands; Relative abundance increased from 0.23/ha in lands with 0.1 to 3% pasture to 0.78/ha in lands with 8 to 28% pasture (3); Occurrence positively correlated with grassland and hay (8); Abundance positively correlated with grassland in 0.5 km landscape (5); Viable population needs 6.5% habitat within 100 km^2 landscape and average patch size greater than 50 ha; Stable population needs 23% habitat within 100 km^2 landscape and average patch size greater than 125 ha; Functional population needs 30% habitat within 100 km^2 landscape and average patch size greater than 150 ha (7)

Species	Relationship with Amount of Habitat in the Landscape
Vesper Sparrow	Abundance positively correlated with crop area in agricultural lands; Relative abundance increased from 0.03/ha in subdivisions with 1 to 20% crop to 0.13/ha in subdivisions with 53 to 84% crop (3)
Le Conte's Sparrow	Abundance positively correlated with grassland in 32 km^2 landscape (5)
American Kestrel	Nest box use positively correlated with amount of pasture and hay within 1 km (9)
Dabbling Waterfowl	Daily nest survival rate greater when 45 to 55% grassland within 41 km^2 landscape compared with 15 to 20% grassland in landscape (11); Nest survival positively correlated with grassland cover within 10.4 and 41.4 km^2 landscapes (12)
Clay-colored Sparrow, Bobolink, Savannah Sparrow and Red-winged Blackbird	Nest parasitism by Brown-headed Cowbird negatively correlated with tree cover within 2 km (13)
Mourning Dove	Abundance positively correlated with amount of grassland and cropland in landscape (14)
Tree Swallow	Number of fledglings and fledging probability decreased with intensification of agricultural land use, but increased with amount of extensive pasture, fallow and hayfield within 5 km (maximum foraging distance) (15)

Citations: (1) Niemuth et al. 2005; (2) Kreuzberg and Lindsay 2010; (3) Kreuzberg and Lindsay unpublished data; (4) Winter et al. 2006a; (5) Quamen 2007; (6) Haroldson et al. 2006; (7) Neave et al. 2009; (8) Ribic and Sample 2001; (9) Smallwood et al. 2009; (10) Jobin et al. 2005; (11) Horn et al. 2005; (12) Stephens et al. 2005; (13) Pietz et al. 2009; (14) Elmore et al. 2007; (15) Ghilain and Bélisle 2008.

In the upper midwestern United States, Meehan et al. (2010) showed that there was a threshold in bird species richness with the amount of high-intensity agriculture on the landscape. They noted that bird species richness in a 25-kilometre square landscape increased linearly with areas of low-input, high-diversity crops (typically pasture); however, when more than 40% of the landscape consisted of high-input, low-diversity crops (typically row crops), bird species richness sharply declined. Species richness also increased linearly with the amount of forest cover on the landscape.

The total area of grassland on a landscape is the key factor in grassland bird richness and abundance. However, while there has been an increase in studies addressing grassland habitat thresholds, the number of studies applicable to the Mixedwood Plains area are limited and based on a small number of species. This currently precludes setting a threshold for grassland extent based on the current populations and richness of grassland birds.

2.4.3 Landscape Configuration, Heterogeneity and Connectivity

> Guideline

Grassland habitat patches should be clustered or aggregated and any intervening land cover should be open or semi-open in order to be permeable to species movement.

> Rationale

There are varying and interacting responses by species to patch size and patch proximity. In the case of bird species, many are not only sensitive to how much habitat exists in the landscape, but also to how that habitat is arranged spatially and the ease of movement between patches. Other birds are sensitive to area independent of the presence of nearby grassland. Similar to the use of small forest patches by forest fauna, in some cases, smaller grassland patches will be used when they are embedded within a landscape with abundant grassland habitat. Clustering grassland patches will benefit species that are influenced by patch proximity, especially for non-bird species of limited mobility, and will also increase the proportion of grassland in the local land-cover matrix. Also, a permeable intervening land cover on the landscape is preferred.

Identifying threshold responses to patch size can be challenging. Ribic et al. (2009) reviewed several studies and identified patch size relationships for 14 grassland bird species that occur within the Mixedwood Plains ecozone (see Table 15); however, many of these studies did not control for local-scale vegetation structure or the amount of suitable habitat available in the landscape, which is likely to confound species-patch size relationships (Bakker et al. 2002; Winter et al. 2006b). Many studies of birds in grassland landscapes have failed to account for the fact that fragmentation (landscape configuration) is confounded with habitat amount (see Fahrig 2003). For example, in mixed-grass and tall grass prairies of southeastern South Dakota, occupancy rates suggest that Sedge Wrens and Clay-colored Sparrows did not discriminate between large and small patches if these were embedded in a grassland-dominated landscape matrix (Bakker et al. 2002). In fact, more individuals were found in small patches within landscapes (400 to 1600 metres surrounding patches) dominated by grassland than in large patches where grassland land cover was low. For shrubland birds in Ohio, densities of most species were greater in patches where greater than 10% of the landscape within a 1-kilometre radius was in early successional stages (Lehnen 2008). In contrast, other species (Grasshopper Sparrow, Dickcissel in mixed-grass prairie and Savannah Sparrow in tall grass prairie) had high occupancy rates in large patches, irrespective of whether or not grassland was dominant in the landscape. This indicates the need to consider the benefit of having habitat patches within a more "open" landscape matrix containing other open patches to accommodate birds that respond positively to being in a grassland matrix. Likewise, in lieu of a grassland matrix, the presence of complementary "open" nearby or adjacent patches will likely have beneficial effects. "Clumping" or aggregating open patches would be a positive action. This could involve seeking opportunities around large "anchor" grassland patches while avoiding fragmentation of forest habitat.

Regional population densities, habitat quality and population factors can also influence patch size relationships, such that area sensitivity may vary regionally (Johnson and Igl 2001). For example, as the densities of Henslow's Sparrow increased within the landscape, breeding birds were found in smaller patches of grassland (Herkert 1994a; Herkert 1994b).

For grassland birds in heterogeneous landscapes, the distance between habitat patches and the quality of the intervening matrix determines whether patches will be occupied or not and the viability of individual patches. For example, patches of grassland separated by more than 15 kilometres of cropland or forest were beyond the dispersal distance of most Bobolinks, and would result in isolated populations (Tews 2008b). Similarly, patches separated by less than 15 kilometres may be appropriate for many other grassland bird species, including Upland Sandpiper and Henslow's Sparrow.

Migratory birds are more mobile than most species groups, making corridors more important for invertebrates, non-avian vertebrates and plants (Gilbert-Norton et al. 2009). When looking at local ecosystems, thresholds for grassland patch connectivity and configuration will be set by the requirements of less-mobile species. This is particularly relevant for native prairie and savannah patches, given that these patches are more likely to provide habitat for a greater diversity of species with limited mobility than surrogate habitats.

The utility of corridors for a variety of habitats has been debated in the literature (Gilbert-Norton et al. 2009). Generally, corridors are seen as beneficial if specific species and habitat attributes are considered, especially in addressing forest habitat connectivity. In regard to grassland habitat corridors, there is limited literature and no clear indication of the benefit of connecting corridors between grassland patches. In view of this uncertainty, another approach that would help serve connectivity needs is to minimize the distances between patches and maintain a matrix that is permeable to dispersing species.

The value of corridors for plants is also unclear. Van Dorp et al. (1997) sees the value of corridors as elusive, while Tikka et al. (2001) concluded that road and railway corridors do serve as dispersal corridors for grassland plants. In terms of animals, narrow buffers may increase their predation while within the corridor (State of Kentucky 2010). In terms of insects, Öckinger and Smith 2007 contend "that corridors do not always have positive effects on insect dispersal," while Haddad (1999) did find that corridors facilitated dispersal in two butterfly species. In a three-year study of inter-patch insect movement, Collinge (2000) suggests that corridors may have some potential to promote movement of individuals, but their use needs to be considered along with species characteristics, landscape context, patch size and environmental variation. Having a complementary matrix and the positive role it plays in dispersal and movement is noted as important (Baum et al. 2004). The negative effect of corridors assisting biological invasions is largely unknown, with Bier and Noss (1998) noting a lack of studies on the subject (and there appear to have been few subsequent studies). Damschen et al. (2006) found that corridors do not directly promote invasion by exotic species. However, Hansen and Clevenger (2005) contend that forest corridor edges and grassland habitats act as microhabitats for non-native species and are more prone to invasion by exotic species than forests, especially if disturbed.

Maintaining a permeable matrix is a more attractive strategy given the potential for predation in corridors and uncertainty regarding the utility of corridors for grassland species and the role of corridors with respect to biological invasions.

Table 15. Evidence for Area Sensitivity in Grassland Bird Species that Occur in the Mixedwood Plains

Species	Occurrence[a]	Reference[b]	Density[c]	Reference[d]	Threshold (Reference)[e]
American Goldfinch	Positive	23			
American Kestrel	Positive	20			Greater than 1000 ha for highest rate of occupancy in nest boxes (20)
Bobolink	Positive Negative	1,2,7,10 13	Positive Variable	3,7,11,12,18 16	Greater than 50 ha for occurrence 50% of maximum (1)
Brown Thrasher	Negative	2			
Brown-headed Cowbird	Negative	7,9	Negative	13	
Clay-colored Sparrow	Positive	7	Positive	7	
Common Yellowthroat			Positive	22	Capture rate higher in 13 to 16 ha patch compared with 4 to 8 ha patch (22)
Dickcissel	Positive	5,8			Patch size did not affect nest success (25)
Eastern Kingbird	Positive	23			
Eastern Meadowlark	Positive	1,2,10	Positive	18	Greater than 5 ha for occurrence 50% of maximum (1); patch size did not affect nest success (25)
Grasshopper Sparrow	Positive Variable	1,2,9,10,14 7	Positive Variable	3,8,9,11,13,18 7	Greater than 12 ha for occurrence 50% of maximum (19); greater than 15 ha for presence (14); greater than 30 ha for occurrence 50% of maximum (1); greater than 100 ha for occurrence 50% of maximum (2)
Henslow's Sparrow	Positive	1,5	Positive	3,6	Greater than 55 ha for occurrence 50% of maximum (1), mean occupied patch size 421 ha (26)

Species	Occurrence[a]	Reference[b]	Density[c]	Reference[d]	Threshold (Reference)[e]
Horned Lark	Positive	13	Positive	13	
Le Conte's Sparrow	Positive	7			
Loggerhead Shrike					Greater than 4.6 ha for presence (21)
Mallard	Positive	23			Nest success higher in larger patch (24)
Mourning Dove	Negative	7	Negative	7	
Northern Harrier			Positive	7,12	Greater than 100 ha for presence (7)
Savannah Sparrow	Positive Variable Negative	1,2 8 7	Positive Variable	3,7,18 16	Greater than 10 ha for occurrence 50% of maximum (2); greater than 40 ha for occurrence 50% of maximum (1); nest success increased with patch size (15)
Sedge Wren	Positive	7,8,23	Variable	7	
Short-eared Owl					Greater than 100 ha for presence (7)
Upland Sandpiper	Positive	2	Positive	3	Greater than 50 ha for presence, greater than 200 ha for occurrence 50% of maximum (2)
Vesper Sparrow	Positive	2			Greater than 20 ha for occurrence 50% of maximum (2)
Western Meadowlark			Positive Negative	4,7,8 15	Greater than 5 ha for occurrence 50% of maximum (19)
White-eyed Vireo			Positive	22	Capture rate higher 13 to 16 ha patch compared with 4 to 8 ha patch (22)

[a] = occurrence in relation to patch size; [b] = associated references; [c] = density in relation to patch size; [d] = associated references; [e] = reference for thresholds

Citations from Ribic et al. (2009): (1) Herkert 1994b; (2) Vickery et al. 1994; (3) Bollinger 1995; (4) Bolger et al. 1997; (5) Winter 1998; (6) Winter and Faaborg 1999; (7) Johnson and Igl 2001; (8) Bakker et al. 2002; (9) Horn et al. 2002; (10) Renfrew 2002; (11) Renfrew and Ribic 2002; (12) Skinner 2004; (13) Dejong et al. 2004; (14) Davis 2004; (15) Davis et al. 2006; (16) Winter et al. 2006b; (17) Winter et al. 2006a; (18) Renfrew and Ribic 2008.

Additional citations: (19) Helzer and Jelinski 1999; (20) Smallwood et al. 2009; (21) Jobin et al. 2005; (22) Rodewald and Vitz 2005; (23) Riffell et al. 2001; (24) Horn et al. 2005; (25) Walk et al. 2010; (26) Herkert 1994a.

2.4.4 Patch Size

> Guideline

Maintain and create small and large grassland patches in existing and potential local grassland landscapes, with an average grassland patch area of greater than or equal to 50 hectares and at least one 100-hectare patch.

> Rationale

Patch requirements in terms of quality, size and surrounding land-cover influence vary by bird species and region. If managing for species diversity, then maintaining a diversity of grassland patch sizes and types will suit the needs of many species, and striving for larger average patch size will accommodate area-sensitive bird species.

Very few studies have examined area sensitivity in grassland birds in Ontario. However, research conducted in other geographical areas suggests that the Upland Sandpiper and Henslow's Sparrow both require very large patches of grassland to establish territories (greater than 50 hectares and preferably larger—as large as 200 hectares for Upland Sandpiper). Tews (2008b) found that viable Bobolink populations were supported in a landscape with average patch size greater or equal to 50 hectares. American Kestrel prefers nest boxes in larger patches (Smallwood et al. 2009). Other grassland birds are potentially area sensitive but have relatively small patch requirements, such as 4.6 hectares for Loggerhead Shrike (Jobin et al. 2005) and 5.0 hectares for both Eastern Meadowlark (Herkert 1994b) and Western Meadowlark (Helzer and Jelinski 1999).

For some species with relatively lower minimum patch-size requirements, there are studies showing positive effects on presence and nesting success when patch sizes are increased, for example nesting success for grassland obligate Savannah Sparrow increased with patch size (Davis et al. 2006).

Winter et al. (2006b) suggested that the specific requirements for the size of a grassland habitat patch will vary among regions, depending on: (1) the quality of the habitat in the patch; (2) the amount of trees and shrubs in the surrounding landscape; and (3) the local predator community. In an unpublished account from New York State, 30- to 100-acre (12- to 40-hectare) patches were recommended to "protect a wide assemblage of grassland-dependent species" while acknowledging a larger patch size is required to protect raptors also (Bittner 2011).

The challenges associated with identifying consistent patch-size thresholds for grassland species suggest that recommendations for patch size should recognize the value of both smaller (see Quamen 2007; Winter et al. 2006b) and larger patches, with average patch area determined by the species with the largest known area sensitivity. Larger patches also provide habitat for a greater number of individuals from a specific species and may support a less variable and possibly more viable local bird community.

Having at least one 100-hectare patch will increase the chances that the most area-sensitive birds will be present in a local landscape, and having a 200-hectare patch will increase the chances of persistence for the most area-sensitive bird species (Table 15).

2.4.5 Landscape Heterogeneity

> Guideline

Some grassland habitat should be located adjacent to hedgerows, riparian and wetland habitats for species that require different habitat types in close proximity.

> Rationale

Grassland species often require a variety of other complementary habitats to complete their life cycles (Justus and Sarkar 2002; Pressey et al. 1993), and many grassland birds require these patches in close proximity to support all life stages (see Table 16). Several waterfowl species, for example, nest in upland grasslands adjacent to wetlands, and bird species that forage in croplands typically nest in adjacent linear features such as hedgerows, fencelines or shelterbelts (Best et al. 1990; Best et al. 1995; Rodenhouse and Best 1983). Other species, such as the Northern Bobwhite or the introduced Ring-necked Pheasant, require a mix of cover for nesting and shelter from predation, as well as grassland and cropland for foraging (Burger et al. 2006).

There has been much discussion over the benefits of isolated or linear landscape features, such as hedgerows. These features can provide sources of food, shelter, nest sites, roosting, foraging sites and song perches for many grassland birds (Best 1983; Cassell and Wiehe 1980; Conover 2005; Conover et al. 2009; Johnson and Beck 1988; Marcus et al. 2000; Martin and Vohs 1978; Smith et al. 2005; Yahner 1982; Yahner 1983). Structural characteristics (height and width) and floristic richness are known to influence species composition of birds (and other taxa). Generally, wider and more diverse hedgerows and those adjoining forest or shrubland patches support more bird species (Best 1983).

Conversely, other studies indicate linear features with woody vegetation may reduce the quality of the adjacent open habitat types (e.g., grassland) for nesting birds, although the impact on density and nesting success may be dependent on the total amount of trees and shrubs on the landscape (Winter et al. 2006b). Nest predation and brood parasitism by Brown-headed Cowbird may be higher in open habitat that is adjacent to areas with woody vegetation. However, the significance of predation can be difficult to predict because the distribution of nest predators in grasslands can be complex (Bergin et al. 2000; Chalfoun et al. 2002b; Winter et al. 2006b).

Table 16. Species Requiring a Mix of Grassland and Other Habitat Types

Species	Open Habitat Type	Other Habitat Types	Notes
American Crow	All open habitats	Forest	Nests in trees, forages in open habitats (1)
American Woodcock	Old field	Shrubland, early successional forest, wetland	Various aged early succession forests on 200 to 400 ha tracts within 1 to 3 km of each other preferred (2) (more open habitats are also used in Ontario)
Baltimore Oriole	Grassland	Forest and treed/shrubby edges	Primarily uses forest, edge and hedgerow habitats (3)
Barn Owl	All open habitats	Barn, tree hollow or nestbox	(4)
Barn Swallow	Grassland	Barn and barnyard preferably with livestock	(5)
Bank Swallow	Grassland	Riparian, or lakeshore, requires nesting substrate	Probability of extinction declined with proximity to grassland (6)
Brewer's Blackbird	Grassland	Perch sites such as fencelines, hedgerows, etc., and water	(7)
Brown Thrasher	Old field	Shrub and shrubby edge (8)	Primarily uses hedgerows and shrubland habitats
Cliff Swallow	Grassland	Windbreaks, trees, water (9), requires structure for nesting	Mean colony size positively correlated with amount of flowing/steady water, negatively correlated with amount of cropland within foraging distance; very large colonies associated with high landscape diversity (9)
Dabbling Waterfowl (American Wigeon, Blue-winged Teal, Gadwall, Mallard, Northern Shoveler)	Grassland	Wetland	90% of grassland nests within 200 m of a wetland, mean 96 m (10)
Gray Partridge	Grassland, cropland	Shrubby edge habitats such as fencelines, hedgerows, etc.	(11)
Northern Bobwhite	Grassland, cropland	Shrubland/forest	(11)

Species	Open Habitat Type	Other Habitat Types	Notes
Red-tailed Hawk	Grassland, cropland	Forest, forested edge	(11)
Ring-necked Pheasant	Grassland, cropland	Scattered small woodlots	(11)
Sandhill Crane	Grassland	Wetland	(11)
Vesper Sparrow	Cropland, grassland	Shrubby hedgerows	Males arrived earlier and had better pairing success where fencerows were shrubby and croplands contained residue (12)
Wild Turkey	Grassland, cropland, old field	Forest	Optimum mix may be 50% forest, 10% row crops, 22% pasture, 13% old field (11)

Citations: (1) Whitney and Marzluff 2009; (2) DeGraaf and Yamasaki 2003; (3) Cadman et al. 2007; (4) Sandilands 2010; (5) Ambrosini et al. 2002; (6) Moffatt et al. 2005; (7) Dunn and Gordon 2007; (8) Vickery et al. 1994; (9) Brown et al. 2002; (10) Henshaw and Leadbeater 1998; (11) Sandilands 2005; (12) Best and Rodenhouse 1984.

2.4.6 Additional Grassland Considerations

Edge density and woody vegetation

There has been much discussion in the scientific literature of the positive and negative effects of edges and woody vegetation on grassland species. In terms of edges, some of the observed variation in minimum patch size may be related to the extent of edge effects in different landscapes. In the past, many researchers believed that early successional or shrubland species were not sensitive to patch size and could live in habitat edges. However, a recent meta-analysis for 17 shrubland species occurring in clear-cuts in a forested landscape demonstrated that all of the species were more abundant in the centre, rather than the edge of patches, and that edge effects were significant for 8 species (Schlossberg and King 2008). Although the study took place within a forested landscape context, similar patterns may occur in fragmented or grassland landscapes, suggesting that more extensive habitat patches are required to support breeding territories.

In grassland landscapes, edges can be pronounced and are often "harder" and more permanent than in forested landscapes. This is because human land uses create linear features with abrupt changes from one vegetation type to another. The role of multiple edges on ecological processes in fragmented landscapes is largely unknown; however, multiple edges increased the magnitude and extent of the effect of edge on Bobolink in Iowa (Fletcher 2005).

Edges also have different microclimates (e.g., light or humidity) than interior habitat, and as a result can affect food supply for various species and can ultimately affect species use, although this has mainly been suggested for forest edges abutting open habitats (Austen et al. 2001; Burke and Nol 2000). Some bird species show a reluctance to cross gaps between a linear non-crop edge and a more hostile open area (such as a crop field) because it exposes them to predation (Bélisle 2005; Bélisle et al. 2001; Bélisle and St. Clair 2001). The quality of edge can also affect bird use of adjacent habitat; for example, some studies have shown that the more complex the edge vegetation, the less likely a bird will venture out into a crop field (Conover et al. 2009). Conversely, other species are attracted to edges such as a fence line because they provide perches from which to forage and vegetation cover for nest sites (Best and Rodenhouse 1984). The National Agri-Environmental Standards Initiative recommends small (1 to 100 hectares) native grassland patches should be surrounded by a greater than or equal to 50-metre buffer comprised of native perennial grasses and forbs, and devoid of woody vegetation and vertical structures (e.g., fences or buildings) (McPherson et al. 2009).

Linear features within or separating open habitat patches, such as hedgerows and shelterbelts, have mixed affects, and what constitutes a barrier or a corridor is not always clear. For example, in a study of the effect of adjacent grassland land cover, Bakker et al. (2002) treated tree-belts or windbreaks greater than or equal to 20 metres wide as barriers and excluded minimum-maintenance roads and fencelines. In terms of use of linear or isolated features, grassland species such as Loggerhead Shrike will use hedgerows and isolated trees or shrubs that might otherwise be considered incompatible with grassland habitat patches. However, edges are avoided by many bird species since they attract many predators, including birds, mammals and reptiles that use linear features as corridors in grassland landscapes (Bergin et al. 2000).

Land management and restoration decisions in relation to edges, in particular linear features such as hedgerows, should consider the needs of grassland and non-grassland species, as well as the provision of other ecological functions, such as soil erosion control. When managing large patches, it should be noted that hedgerows, shelterbelts and other isolated and linear woody features do not necessarily constitute a break within a patch nor generate a negative edge effect. These features should generally be retained in the face of other concerns.

Timing

Replicating natural seasonal habitat cycles can help address the temporal disconnect between human and wildlife land uses. Temporal heterogeneity in grassland landscapes needs to be maintained through periodic disturbance, and patches of tall, medium and short grasses need to be represented and stand in place until mid-July for use by nesting birds. Other grassland habitats such as old field need to be renewed on a longer cycle.

Small changes in the timing of operations could contribute greatly to maintaining population viability and improving habitat for birds. Mowing, for example, can create a habitat sink or ecological trap (an apparently productive area that actually produces a net loss to the species). This occurs in hay fields unless the mowing date is adjusted (e.g., after July 7, Nocera et al. 2005; after July 15, Dale et al. 1997; Quamen 2007). Mowing before these dates can result in as much as 94% mortality in nestling and juvenile birds (Bollinger et al. 1990; Dale et al. 1997). Herkert’s (1998) recommendations to adopt rotational management combined with avoidance of cutting before mid-July may provide one way of reducing the impacts. The National Agri-Environmental Standards Initiative guidelines (Neave et al. 2009) outline many best management practices for grassland habitats.

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