Restoring a wetland complex for amphibian populations, south Okanagan Valley, British Columbia, Canada (2003 to 2014)

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Statement of Contributions

This thesis contains four articles that are collaborative efforts of multiple researchers that will be submitted into peer-reviewed journals. The thesis Introduction (Chapter 1) contains elements of a review article to be submitted to an environmental planning journal such as Ecological Applications. The second paper (Chapter 2) is a review article to be submitted to Conservation Letters. The third paper (Chapter 3) will be submitted to Canadian Field Naturalist or the Journal of Biodiversity and Conservation. The fourth paper (Chapter 4) will be submitted to Restoration Ecology or Agriculture, Ecosystems and Ecology. All materials in composition of the original journal articles provided in the thesis are the sole production of the primary investigator listed as first author. The specific contributions of the co-authors to each paper are listed below.

The written portions of the review articles (Chapter 1 and Chapter 2) was completed in its entirety by Sara Ashpole and edited by co-authors Christine Bishop and Steven Murphy.

The written portions of the manuscripts (Chapter 3 and Chapter 4), including figures and tables, was completed in its entirety by Sara Ashpole and edited for content and composition by co-authors Christine Bishop and Steven Murphy. The statistical analysis in Chapter 3 is a direct contribution between Sara Ashpole and Shane de Solla. The field methodology and data collection was developed and completed by Sara Ashpole.

The undersigned are in agreement with the evaluation of contributions and roles of the authors stated in the Statement of Contributions.

Stephen D. Murphy	Date
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Abstract

The arid south Okanagan Valley is a highly anthropogenic landscape experiencing intense development pressure from agriculture and urban expansion. Ecologically, wetland and riparian habitat loss now exceeds 84% of what existed since the 1800s. Based on these conditions, I tested whether species richness, distribution, and relative density of native herpetofauna among 108 wetlands surveyed during 2003 to 2006 would show significant differences among sites as defined by their land-use characteristics. I identified seven land-use stressors: water withdrawal or discharge; infilling or shoreline modification; burn pile / garbage dumping; non-native invasive species; agricultural input (e.g. pesticides, herbicides); nutrient input (unrestricted livestock, turf fertilization); and artificially constructed sites. At least one of the seven identified land-use stressors was present at 88% of sites and 74% of sites experienced nutrient inputs. The highest overall frequency of stressors occurred at agricultural sites. And yet, these agricultural sites breeding habitat value with the highest species richness of native herpetofauna and some of the highest observed densities of species early life stages. Despite repeat surveying, more than two-thirds of sites had less than two herpetofaunal species detected annually. In response to the apparent ecological degradation based on field observations, a collaborative stakeholder approach was initiated to increase the quantity and quality of lowland wetland habitat. The approach used was landscape ecological restoration, i.e. reconnecting known amphibian-breeding sites with constructed and/or enhanced small ponds. The prior herpetofauna monitoring data (2003 to 2006) determined both ecological and management based strategic locations: 1) proximity to known herpetofaunal breeding locations, 2) distance to adjacent water bodies, 3) distance to roadways, 4) historic wetland infilling, or 5) partnership with local conservation authorities. Habitat enhancement outcomes initiated as part of my research (N_{total} = 21 sites) included 10 newly constructed ponds, enhancement of 8 re-contoured ponds after historic infilling, and invasive non-native predatory species removal at 3 sites (2006 to 2011). Project ponds were monitored annually (2007 to 2014) for all life stages of herpetofauna. Over this eight year period, metamorphic success for the Great Basin spadefoot (*Spea intermontana)* (13 sites) and the pacific chorus frog (*Pseudacris regilla*) (N = 7 sites) populations has been observed. Enhancement and construction of ponds in the lower valley roughly doubled the number of available discrete breeding ponds within the study area from 13 to 31 ponds and through voluntary stewardship engaged landowners. Whether enhancement and construction of ponds have aided species recovery is unclear, because it can take multiple species generations and habitat protection to establish whether there has been a permanent ecological restoration; however the data on colonization provides early supporting evidence that some species are recovering. The project results support a “build it and they will come” action; this approach likely works because there was severe decline in available breeding habitat, hence any improvement tends to provide an initial increase in some widely distributed populations of herpetofauna. Nonetheless, the ongoing lack of critical upland habitat needed by amphibian species poses a significant threat to species movement and long-term population success. Planning and management challenges remain, namely enforcement of wetland protection measures and moving beyond like-minded collaborations and towards targeted stewardship of less motivated persons.

Key words: anthropogenic stressors, constructed wetlands, amphibian diversity; ecological planning and management; agroecosystems; restoration ecology

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> "*It's not pining, it's passed on! This parrot is no more! It has ceased to be! It's expired and gone to meet its maker! This is a late parrot! It's a stiff! Bereft of life, it rests in peace! If you hadn't nailed it to the perch, it would be pushing up the daisies! It's rung down the curtain and joined the choir invisible. This is an ex-parrot!*" John Cleese, 1969.

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1.0 INTRODUCTION

This thesis examines how restoration ecology and ecological planning address degradation of ecosystems in professional practice. Background knowledge (Chapter 1) related to agricultural lands and amphibian species ecology and wetland restoration is provided in support of the proceeding manuscripts examining wetland monitoring (Chapter 3) and restoration activities (Chapter 4) in the south Okanagan Valley, British Columbia with a primary synthesis and discussion of the thesis in Chapter 4 (section 4.4. and 4.5).

First, I briefly review ecological planning theory and followed by a synopsis of eight common restoration challenges to achieving ecological integrity as a primary goal of ecological planning. In response to these challenges, I outline how to avert or repair ecosystem degradation via professional practice of ecological planning and restoration ecology (*defined in* Box 1).

> Box. 1 **Restoration Terminology.** While "restoration ecology" tends to focus mostly on experimental and innovative approaches from the natural and physical sciences, "environmental restoration" focuses on the practitioner-based approaches needed to restore a site to some reference or at least acceptable state. Further, environmental restoration encompasses function and social factors such as aesthetics, economics, governance, and policy (Gross, 2007). There is a logical progression from the science of restoration ecology to the practice of environmental restoration to the implementation of ecological planning. The phrase "**restoration**" as used in this thesis encompasses both rehabilitation and enhancement, and is used to simply describe in a broad sense ecological efforts to address habitat degradation. Rehabilitation and enhancement refer to the progression of a former ecosystem structure and function to a new desired state (Grenfell et al., 2006). Restoration generally refers to the attainment of the former ecosystem structure and function prior to anthropogenic disturbance. Restoration may also encompass systems that have been lost entirely and are being reconstructed (Grenfell et al., 2006). **Ecosystem Integrity** is defined as "the state or condition an ecosystem that displays the biodiversity characteristics of the reference, such as species composition and community structure, and is fully capable of sustaining normal ecosystem functioning" (Society for Ecological Restoration, http://www.ser.org). The phrase "**landscape**" implies both a spatially explicit geographic area with structure and ecological processes.

Environmental restoration can be transdisciplinary in approach, and may emphasize that humans are a part of the ecosystem with a responsibility to steward and foster processes that facilitate and accelerate the recovery of ecosystems with respect to environmental sustainability and ecological integrity. Environmental sustainability refers to the relative assessment and consequential trade-offs between productive land-use and the maintenance of long-term ecosystem function of the environment. Measures taken to restore ecosystem degradation contribute to ecological integrity of the system. The challenge for the concept of ecological integrity is to provide the means to distinguish between responses that represent improvements in the quality of ecosystems and those that represent degradations. Environmental restoration is often subsumed into the wider field of ecological planning where the goal is to provide policies and practices that conserve, maintain or repair ecosystem services. The concept of natural values is intrinsic in ecological planning and in attaining many ecosystem services (Kass et al., 2011; Pressey et al., 2007). Natural values are shaped by widely shared societal meanings that humans attach to the natural environment and can be associated with one's own beliefs and also in monetary values attached to the environment (*see* example Bendor, 2009).

The role of ecological planners, as an interdisciplinary team member, is to ensure that restorative environments are protected and to design practices that achieve explicit quantitative objects for a desired state of the environment (Han, 2003; Pressey et al., 2007). This Introduction chapter focuses on the current challenges of environmental restoration encountered by practitioners and the supporting evidence of restored structure and function of degraded landscapes through environmental restoration and ecological planning. Moreover, the conceptual and practical basis supporting environmental restoration is discussed.

1.1 Planning theory and practice as applied to conservation planners

The Rational Comprehensive planning model (cf. Mitchell, 1997) can be best described as the government's role in providing policies and legislation to protect Canadian biodiversity. The process of federal policy development and goal setting in Canada incorporates scientific theory, expert contribution, and a centralized decision making system. However, the implementation of federal policy to protect Canadian biodiversity is not a centralized process. The Rational Comprehensive model has a distinct stepwise approach that mirrors the governmental approach to assessing endangered species and establishing species strategies. Rational Comprehensive planning is centered on six defining steps: 1) problem formulation, 2) establishing goals and objectives, 3) identifying alternative means, 4) assessing options against criteria, 5) selecting and implementing a solution, 6) and monitoring and evaluating. The enforcement of federal and provincial policy reflects a 'top-down' management approach characteristic of the Rational Comprehensive model. The Rational Comprehensive model is a valid approach, but loses practicality in achieving measurable environmental benefits. The public by nature will not follow an approach if they perceive it as an unjust cause or an impractical solution to environmental problems that impact their lives or if they do not trust the government. Canadian legislation aimed to protect biodiversity is poorly suited to address complex landscapes, varied regional goals, or demands of the economic sectors. To address the shortcomings of policy and legislation, conservation planning and ecological restoration across Canada is often implemented using a Transactive Planning Model (described in Mitchell, 1997).

Restoration planning that draws people into the process enables capacity for proactive practice and 'aftercare' within a meaningful place or space and allows participants and planners to learn in real time from mistakes made (Friedmann, 1993). Subsequently, successful conservation initiatives and restoration projects address habitats where people and wildlife co-habit as a single working unit (*see* Ancrenaz et al., 2007; and Goosen et al., 2007). Governmental stewardship programs implemented by bipartisan-neutral environmental groups provide a non-confrontational 'bottom-up' approach to conservation planning. Utilizing and developing community and stakeholder support is critical for both the initiating incentive for restoration action and also the implementation and long-term commitment required for restoration success. Stewardship programs, in addition to relying on expert knowledge and decentralized decision-making, are inclusive of the people affected in the process of conservation. The negative impacts of degradation to community and ecosystem health can provide strong motivation for change and participation in restoration action. The inclusion of affected individuals provides local expertise and necessary context to conservation planning and ecological restoration. The partnerships nurtured during the process of stewardship-driven planning are valued and the role of the planner becomes more of a facilitator representing stakeholders to develop widely accepted plans (Mutshewa, 2010). Fostering successful stewardship relationships requires detailed attention to the planning process, rather than merely focusing on short-term outcomes.

The Transactive planning approach requires greater time investment on the part of the conservation planner; however the viewpoints of affected individuals may not be equally represented. In the South Okanagan Wetland Restoration (SOWR) example (presented in Chapter 4), it is apparent that some ethnic groups are under-represented during the consultation process making conservation actions more challenging in practice than in theory. Specialist conservation groups who receive federal funds for stewardship may implement an approach closely reflective of an Advocacy Planning Model (as described in Mitchell, 1997), whereby an organization represents a broader societal interest possibly at the expense of other groups involved. Rational Comprehensive, Transactive, and to a lesser extent Advocacy planning, encompass the current Canadian theoretical approaches to conservation planning for biodiversity protection. No single theoretical planning approach is without challenges or imperfections. However, established theories provide a useful framework for both theoretical and practical implementation at the various levels of conservation planning and ecological restoration in Canada.

1.2 Ecological planning and environmental restoration in practice

Restoration is a process to repair degraded ecosystems in a matter of years compared to the natural processes that would normally take decades to develop or be effective (Hilderbrand et al., 2005). The responsibility of preserving, protecting, managing, and restoring ecosystems is shared among governmental and non-governmental agencies, the private sector, and members of the public. Environmental literature addresses the approaches taken in restoration ecology as a science, as environmental restoration, and as ecological planning. Yet the merging of practice and theory presents an obstacle for ecological planners. The failure to address restoration project shortcomings and evaluation hinders the progress of practitioners and the process of restoration. Until solutions are addressed and implemented into ecological planning, resources will be wasted and beneficial effects of restoration will be reduced. To support restoration practice, eight widely acknowledged restoration issues or practical problems are presented from the ecological planning literature along with appropriate recommendations (Fig. 1.1; Clewell and Rieger, 1997; Quon et al., 2001; Bernhardt et al., 2005; Miller and Hobbs, 2007; Palmer et al., 2007; Pressey et al., 2007). The challenges presented in ecological planning are rarely independent and tend to cumulate into greater challenges that affect the ongoing success of a project or ecosystem plan.

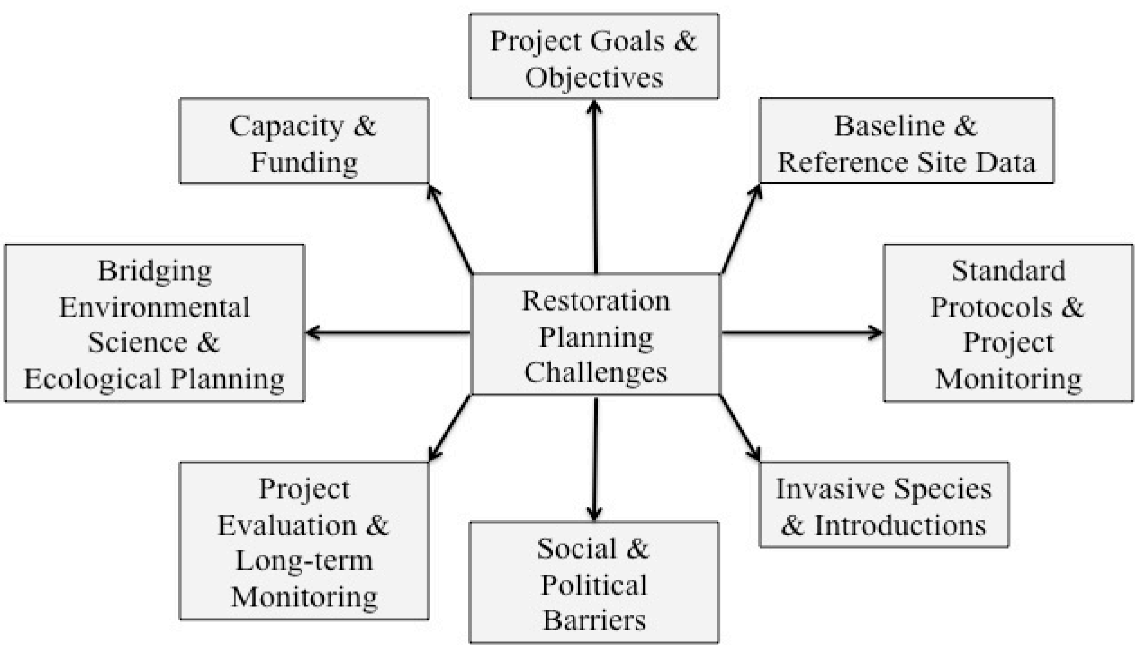

Figure 1.1. Eight common project challenges, characteristically connected clockwise from top, experienced by restoration and conservation planners.

1.2.1. Project goals and objectives

Challenge: Designing measurable goals and objectives that are realistic within time constraints that incorporate wide scale area objectives.

Recommendations: Practitioners need to set ecologically defensible and measurable goals and objectives that correspond to the desired ecological state of a site. Goals and objectives should align with regional priorities and move away from a narrow scale project perspective (Quon et al., 2001; Hilderbrand et al., 2005). Meaningful goals and objectives are achieved by the integration of long-term objectives and the utilization of information resources while considering the values of participants and stakeholders (Cooper et al., 2007; Miller and Hobbs, 2007). Funding and permitting agencies should require the submission of the initial project design with ecologically measurable goals and subsequent ongoing reports demonstrating goal achievement (Hilderbrand et al., 2005). A progressive approach to project goal setting must move beyond quantitative targets for ecological attributes, such as vegetation density or hydroperiod, but towards considering ecological capital. Connectivity and variability are measures of ecological capital that are likely to improve the ecological resilience of restored systems (Hilderbrand et al., 2005; Elmqvist et al., 2003).

1.2.2 Baseline and reference site data

Challenge: Project success is difficult to establish unless baseline site conditions, pre- and post-restoration conditions, and comparisons with multiple reference sites are assessed.

Recommendations: Few ecosystems have been studied extensively in terms of biotic and abiotic parameters, particularly in terms of their responses to natural disturbance. Ecological studies must focus on acquiring baseline data where information is lacking, particularly regarding species inventories, habitat requirements, and their functional attributes (Clewell and Rieger, 1997; Hobbs and Harris, 2001). In most cases, establishing or recreating a site to a pre-disturbance state is not viable or practical for ecological practitioners. Selecting an appropriate time frame as a reference point for restoration comparisons is not easy, and knowledge of conditions and rates of landscape change may not be known, particularly prior to postcolonial European settlement when written documents and original land surveys were generated. Recreating pre-disturbance conditions in developed landscapes is often in conflict with the interests of stakeholders (Hilderbrand et al., 2005). Most often the assessment of baseline conditions happens just prior to the initiation of restoration activities. Pre-condition area data should be compared to post-restoration conditions and reference sites. Multiple reference sites should be examined in order to adequately account for natural dynamics or heterogeneity of the physical environment (Clewell and Rieger, 1997). A progressive restoration approach involves moving away from the often unrealistic single state end point for an ecosystem (Hilderbrand et al., 2005). Ecosystems should be considered dynamic, with the main focus should center on repairing damaged ecosystem functions to the greatest extent possible (Hilderbrand et al., 2005).

1.2.3 Standard protocols and project monitoring

Challenges: Environmental restoration as a practice and restoration ecology as a science lacks standardized, replicable, and validated monitoring and standard protocols, and correspondingly standardized reporting procedures.

Recommendations: Most restoration projects are inadequately monitored, which leaves little opportunity to evaluate the methods used (Clewell and Rieger, 1997; Palmer et al., 2007). A lack of standardized restoration methodology results from uncertainty in the predictability of complex ecological interactions within unique landscapes and possibly a failure to publish the efforts of failed projects. A tendency for practitioners is to adopt monitoring methods or protocols without questioning their efficacy or validity, thus perpetuating repeated mistakes and increasing the financial cost of the project (Clewell and Rieger, 1997; Quon et al., 2001; Pressey et al., 2007). A reevaluation of earlier restoration projects should be examined for the purpose of discovering long-term outcomes of applied methods (Clewell and Rieger, 1997). The evaluation and reporting of standard protocols should be incorporated into the adaptive management plan to allow for mid-course corrections and for comparisons of efficacy among projects (Clewell and Rieger, 1997; Quon et al., 2001). There is a growing need to develop statistically dependable and cost effective methods for the successful evaluation of ecosystem function (Palmer et al., 2007). The incorporation of appropriate technological applications, such as Geographic Information Systems (GIS), aids decision making for practitioners. Effective monitoring protocols should provide supplementary information on relevant aspects of the project. To provide clear justification of actions, standard monitoring protocols must move beyond following a simple protocol towards reasoning: this is achieved by incorporating the current ecological state, background, nature and context of the project (Palmer et al., 2007).

1.2.4 Non-native invasive species and species introductions

Challenges: Management of non-native invasive species in restoration projects may exceed the scope and length of a project. Non-native invasive species may be unintentionally overlooked if project goals and objectives do not specifically address their potential impact. An emerging issue for restoration practitioners is the unintentional release or planting of organisms without proper consideration for long-term consequences to local biodiversity.

Recommendations: Ecological planning may promote unintentional introduction of a non-native invasive species. Impact assessment of potential non-native invasive species on the existing restoration goals and objectives should be incorporated into an adaptive management plan. Specifically, restoration plans should address access to high-risk habitats or high-risk pathways susceptible to invasion. Indirect management (e.g. education campaigns) of the ecological community may be more effective than the removal of the invader species alone. Successful species management focuses more on community level recovery and less on traditional approaches, such as reducing invader density (Buckley, 2008). Conservation and restoration strategies need to consider the evolutionary adaptation of species introduction, as planning decisions may facilitate or hinder the persistence of biodiversity (Rice and Emery, 2003). Generally, practitioners have ignored genetic processes and many non-native or non-local species have been unintentionally, and perhaps more frequently intentionally, introduced at project sites. Precautionary principles suggest that ecological planners incorporate local species when planting or releasing animals. However, the emerging science of genetic restoration ecology is exploring the manipulation of genetic structure of floral populations to maximize the adaptive potential of restored populations (Rice and Emery, 2003).

1.2.5 Social and political tools

Challenge: Ensuring significant and enduring social and political support for projects while considering both ecological and human values is essential.

Recommendations: Bridging social and political interest with restoration provides understanding of the natural values and services of the environment, such as wildlife habitat, clean water and human well-being. Social barriers (e.g. level of education, cultural attributes to nature) can be significantly reduced if stakeholders are included early in the consultation process. Providing transparent project goals and adopting a communication strategy will aid in implementation and conflict resolution with stakeholders (Quon et al., 2001). Citizen initiated projects, citizen science, and participatory research require a supporting infrastructure for collaboration with researchers and decision makers. Restoration projects initiated by local citizens, or community projects which respond to the needs of the community, promote knowledge and awareness and foster collaborations between stakeholders (Cooper et al., 2007). An understanding of policy in context of the site and how projects are officially or unofficially supported, managed, and protected is critical for insuring both public and political support (Quon et al., 2001; Cooper et al., 2007). Jurisdictions and agencies globally are experiencing reduced budgets and staffing which heighten the need for community buy-in. Additional amendments to laws in some areas are undermining landscape scale ecological planning (De Francesca, 1997 as cited in Quon et al., 2001). Even when restoration projects have strong public and stakeholder support, without economic and political commitment or the legal mechanism available to support ecological planning decisions (Quon et al., 2001) restoration projects may be incomplete or fail.

1.2.6 Project evaluation and long-term studies

Challenge: Few restoration projects incorporate or due to constraints are unable to conduct adequate pre-, post-, or long-term monitoring that effectively evaluates project success. In cases where evaluation is conducted, few projects document outcomes accessible to practitioners.

Recommendations: Restoration project design should incorporate evaluation-based goals, use acceptable methodologies, and secure accessibility to long-term funding. Determining the degree of restoration success can be difficult, particularly in terms of ecosystem function that is often deemed subjective (Clewell and Rieger, 1997). Long-term studies are critical in determining the persistence of faunal and floral responses to restoration efforts, and in establishing the biological variation of sites. Practitioners need to utilize and publish peer-reviewed documents, or through other accessible means identify factors contributing to the success or failure of their project (Quon et al., 2001; Palmer et al., 2007). As more long-term data and expert knowledge become available, practitioners can incorporate research findings into their projects and refine their own research methods and evaluation measures (Cowling et al., 2003).

A major challenge among practitioners and is constructing a user-friendly common platform database that allows information sharing without the often-conflicting need for proprietary license. Work on this proceeds (e.g. Ontario Ministry of Natural Resources Online Biodiversity Database; Conservation Data Center, Environmental directories) but the conflicting goals of sharing, secrecy, involvement of community members, and data quality control will make swift implementation unlikely, aside from any technological platform challenges.

1.2.7 Bridging environmental science with ecological planning

Challenge: Ecological planning needs to develop and keep pace with restoration ecology as a science and environmental restoration as a practice.

Recommendations: There is a recognized gap between environmental restoration principles in the literature and the principles implemented at sites by practitioners (Cairns, 1993; Quon et al., 2001; Palmer et al., 2007; Pressey et al., 2007). The scientific understanding of restoration is becoming increasingly sophisticated; yet little integrative understanding of ecosystems is translated to integrative management that encompasses multiple processes simultaneously by practitioners (Cairns, 1993; Pressey et al., 2007). Environmental scientists need to communicate more effectively and transparently with practitioners and stakeholders by actively explaining science and by engaging in long-term collaborations to promote effective implementation (Gross, 2007; Pressey et al., 2007). To facilitate the dissemination of information, written records need to be made accessible to other practitioners and peers (Palmer et al., 2007). A progressive decision making approach, such as the National River Restoration Science Project (NRRS, USA), incorporates the use of an accessible database where project plans and subsequent outcomes are logged (Bernhardt et al., 2007). Accessible databases aid practitioners in coordinating regional goals, in implementing appropriate methodologies, and in conducting project evaluation (NRRS database, Bernhardt et al., 2007; Freshwater habitat database, Katz et al., 2007).

1.2.8 Capacity and funds

Challenge: Insufficient or non-persistent funds or the capacity by participants and stakeholders to complete projects in entirety can seriously undermine the successful completion of a project.

Recommendations: Securing funds for restoration activities is one of the greatest limitations experienced by practitioners (Quon et al., 2001; Palmer et al., 2007). Practitioners need to make decisions and be conscientious about how to allocate limited funds and resources to achieve maximum project benefit. Financial constraints commonly occur when project consultants or sponsors meet the minimum needs of projects or when there is a low priority by policy makers for implementation (Murphy et al. 2007; Palmer et al., 2007). Project design requires clear priorities and measurable goals with a proper funding mechanism to address each priority (Palmer et al., 2007). Restoration strategies that have goals incorporating both a biological and an economic perspective are more likely to receive funds required for project implementation (Cowling et al., 2003).

The process of designing and implementing a restoration project is complex. Practitioners face ongoing challenges stemming from incomplete biological information or poor project support from policy makers. The eight identified challenges are experienced by practitioners and environmental scientists and are intended to embrace the breadth of opportunities that exists for the advancement of restoration ecology and for the field of environmental restoration. The goal of practitioners is to incorporate as many of the presented recommendations as possible into ecological planning. Environmental restoration should grow into a realm of an established technology (Clewell and Rieger, 1997) in order to repair the ecological damage caused by humanity.

1.3 Achievements in environmental restoration

Restoration actions can have measurable and direct benefits for endangered species recovery, including predator eradication (Towns et al., 2001), recreation of a diverse grassland (Bullock et al., 2007), and the increase of landscape connectivity for avifauna (Aerts et al., 2008). Interpreting complex relationships among species and habitat types is important in establishing the conservation implications of highly degraded landscapes. Single species strategies with a narrow focus fail to address the needs of complex ecosystems. Financially feasible and ecologically sensible projects address restoration of entire systems. Many published restoration projects deem success (Table 1.1) while acknowledging concerns raised regarding the eight main shortfalls of restoration projects discussed in this paper. Some projects may neglect or overlook major errors, or the course of project progression at the time of publication may not include the examination of the final data. Regardless of discipline, the 'File Drawer Problem' of risking type II errors and the inherent preference for individuals to publish success and avoid failure contribute to the loss of many relevant findings (Bauchau, 1997). Bernhardt et al. (2005) examined more than 37,000 project records on American river restoration and identified the failure of more than 90% of the projects to monitor or report project findings. Consequently, more than $1 billion per year is spent in the US on river restoration without any form of project evaluation. The Bernhardt et al. (2005) publication prompted an intensive interviewing campaign of practitioners on their motivation, implementation, and project assessment, and also brought about the launching of a national program on strategic monitoring (NRRS, Bernhardt et al., 2007). Future assessment of the success of the national strategic program is an anticipated and invaluable model for the field of environmental restoration, as well as for worldwide standardization.

Table 1.1. A global literature search on the environmental restoration and planning literature (using ISI Web of Knowledge®) provide several thousand published restoration projects carried out within that past 20 years. The restoration projects identified in the literature are at various stages of development, implementation, and monitoring.

The type and degree of degradation varies widely, including invasive species	Restoration projects have examined many floral communities and almost every class of vertebrate species	Evidence of restored structure and function of the degraded landscape deem ecological success	The response time for ecosystem structure and function to be restored can be highly variable

• small mammals (New Zealand, Towns et al., 2001)
• flora and grasses, (England, Bullock et al., 2007; Cox and Allen, 2008; Ethiopia, Aerts et al., 2008)
• natural resource extraction and toxic contamination (radio-nucleotides, Burger et al., 2007; mining, Australia, Nichols and Grant, 2007; Majer et al., 2007; Nichols and Grant, 2007)
• agricultural (grazing suppression, Argentina, Leynaud and Bucher, 2005, Burtin et al, 2009; hay farms back to salt marshes, USA, Able et. al., 2008)
• fire suppression (Converse et al., 2006)
• plethora of general habitat improvements resulting from human development
  • river restoration, Bernhardt et al., 2007
  • poor water quality, Kattz et al., 2007
  • infilling, Kondolf et al., 2007
  • re-vegetation Vesk et al., 2008
	• reptiles (Towns et al., 2001)
	• small mammals and pine forests (Converse et al., 2006)
	• tidal and salt marsh vegetation assemblages (Konisky et al., 2006)
	• grasslands (Bullock et al., 2007)
	• nutrient cycling (Nichols and Grant 2007)
	• invertebrates (Majer et al., 2007)
	• multi-species (fish, benthic infauna, reptiles, invertebrates, aquatic vegetation, Able et. al., 2008)
	• avifauna (Aerts et al., 2008;)
	• sage scrub and exotics (Cox and Allen, 2008)
	• vegetation assemblages (Vesk et al., 2008)
		• changes in vegetation density and diversity (structural) or the nitrogen-fixing capabilities and cycling processes in floral assemblages (functional) (Converse et al., 2006; Grant et al., 2007, Purrenhage and Boone, 2009)
		• projects that focus on restoring wildlife species typically measure structural (re-colonization, distribution and abundance of species) and functional (feeding, growth, reproduction) aspects for evaluation (Nichols and Grant, 2007; Able et al., 2008)
		• establishing functional measures of restoration success characteristically require years of long-term monitoring and adaptive methodology
		  • 25 years, Leynaud and Bucher, 2005
		  • 8 years, Bullock et al., 2007
		  • 20 years, Grant et al., 2007
		  • 30 years, Majer et al., 2007
		  • 9 years, Able et. al., 2008
		  • 5 -130 years, Vesk et al., 2008
			• responses to restoration may be immediate, such as faunal response to flooded inter-tidal surface (Abel et al., 2008), or may take hundreds of years to establish, such as significant tree girth growth (Vesk et al., 2008)
			• few studies refer to the time required to measure maintenance of biological relationships in an ecosystem (Bullock et al., 2007)
			• establishing the maintenance of biological relationships requires monitoring after structure and function have become measurable. Ensuring maintenance of biological relationships reinforces that restoration is treated as an ongoing program, not as a discrete project with a terminal end date

1.4 A practical basis for ecological planning and environmental restoration

Though the fields of ecological planning and environmental restoration are linked in a 'motherhood-like' structure, as such relatively uncontroversial, there is not uniform support for either discipline. The financial costs of restoration can be high, hence it may not be easy or immediately feasible to restore severely modified sites. Some environmental practitioners believe that ecosystems are able to self-repair. The amount of disturbance a system can absorb, while still remaining within the same state, is known as ecosystem resilience (Elmqvist et al., 2003). Policymakers and politicians often believe that there is greater practicality in the preservation and protection of existing habitats, rather than in the replication of the pre-disturbance conditions (Cairns, 1993). Nature reserves are valuable to humankind and to their inhabiting species and should exist in all regions of the world. Reserves should be selected based on a highly complex set of biological and physical processes (Cowling et al., 2003; Pressey et al., 2007). To neglect this complexity is to jeopardize the persistence and preservation of many processes which will result ultimately in the poor functioning of reserves.

Increasing human expansion, progressive environmental degradation, and escalating demands on ecosystem services (Cairns, 1993; Hilderbrand et al., 2005; Miller, 2006; Palmer et al., 2007; Pressey et al., 2007) may exceed any strategy that relies on preservation and protection of nature reserves (Wiersma et al., 2004). The number or area of nature reserves is minimal, fragmented, and infrequently located in biologically significant areas (Gurd et al. 2001; Deguise and Kerr, 2006). Nevertheless, ecological planners and conservationists have a responsibility to preserve and protect even small reserves in urban areas that may be suitable for some species (Miller, 2006; Crossman, et al., 2007) or may significantly contribute to system function, environmental heterogeneity, and local biodiversity. The preservation and protection of reserves insufficiently address today's environmental issues, such as air and water quality. Ecologist may share the belief that greater practicality exists in the rehabilitation or reconciliation (Rosenzweig, 2003), rather than the exclusive restoration of ecosystems. Humankind needs to reduce the demands on ecosystem services in order to better focus on our ability to create, restore, and enhance ecosystems globally (Hilderbrand et al., 2005). Practical and progressive planning should aim for a 'no net loss' strategy by counter-balancing impairment with restoration in addition to the protection of reserve systems. Policies that implement protection and ecosystem restoration that significantly exceed rates of loss can achieve the 'no net loss' strategy (Cairns, 1993). Restoration activities or gains are not functionally equivalent to ecosystem losses (Hilderbrand et al., 2005), even if an aggressive no net loss strategy is implemented. While repairing environmental damage is considered an immense task, the prospect of permanent loss of ecological services and amenities is a far greater cost to humankind.

1.5 Conclusion: A transformative approach to the restoration of degraded ecosystems

Restoration ecology, environmental restoration, and ecological planning all address the degradation of ecosystems in professional practice. Environmental restoration is practical and it is essential. The field of environmental restoration as a science and a transformative practice is rapidly advancing. The past few decades have contributed a plethora of international research that provides lessons for successful implementation of environmental restoration. There are many examples of achievements in restoration success throughout the world that are represented in the wider published environmental restoration literature (Towns et al., 2001; Leynaud and Bucher, 2005; Converse et al., 2006; Konisky et al., 2006; Bullock et al., 2007; Grant et al., 2007; Majer et al., 2007; Nichols and Grant, 2007; Able et. al., 2008; Aerts et al., 2008; Cox and Allen, 2008; Vesk et al., 2008). Many represent meta-reviews, have been accepted through critical peer review and represent many years of research and evaluation. The achievements in restoration illustrate the fundamental importance of addressing the eight challenges experienced by practitioners. If solutions, particularly related to goal setting and evaluation, are not incorporated into practice then fixing ecosystem degradation will not likely work. Despite the challenges of environmental restoration practitioners, the achievements of successful restoration projects inspire and drive the ecological community to surpass and overcome the limitations. Success, however, is never guaranteed and the key issues for future research and practice will be standardization, reporting, and evaluation. Unless political action fully supports scientific innovation in restoration ecology responsible ecological actions cannot take place. An immediate intervention is needed to transform our current way of thinking and our present approach to environmental degradation towards restoration. The failure to conduct environmental restoration will result in the ultimate loss of species biodiversity and will lead to irreparable damage to existing ecosystems.

2.0 A REVIEW OF AMPHIBIAN AND WETLAND CONSERVATION PLANNING IN AGRO-ECOSYSTEMS

2.1 Wildlife and habitat biodiversity in agricultural landscapes

The greatest global risk to wildlife over the history of human expansion is the conversion of habitat to other uses, namely agriculture and urban development (Bulte and Horan, 2003). The preservation and restoration of biodiversity in agricultural landscapes is a pressing conservation issue for the twenty first century. Land committed for resource production by human society occupies a substantial proportion of the terrestrial environment throughout the globe. The persistence and capacity of many species to survive is related (often negatively) to land-use in a human dominated landscape. Since the 1600s, approximately twenty-one percent of documented species extinctions have been directly attributed to habitat loss (World Conservation Monitoring Centre, cited in Bulte and Horan, 2003). Although many farming practices damage the environment, the effects of agricultural practices can be mitigated against by conscientious land-use and policy approaches. Agricultural practices having a positive effect on biodiversity include conservation tillage, maintaining hedgerows (Schuler et al, 2013), managing livestock grazing on natural grasslands (Burton et al., 2009), enhancing buffers and riparian areas (Plieninger et al., 2006). Alternative approaches maintain traditional farming practices and less mechanized land-use methods as seen in organic farm settings (Plieninger et al., 2006) or strategies involving land-sparing (Mattison and Norris, 2005) or conservation-reserve planning (Smith et al., 2012)

The past two decades have seen the birth of several policy initiatives directly bearing on Canadian agriculture and biodiversity. The measurable influence of key agricultural polices through government, non-government, and agriculturalist partnerships are a step towards ecosystem maintenance and recovery and address the shortcomings (e.g. Species at Risk Act) at the Federal protection level. The current dissertation analysis is based on agricultural and environmental stewardship policy and conditions of the last 12-yrs. However, since the restoration research was initiated in 2006 (*see* Chapter 4) an uncertain and declining environmental shift has occurred with resource development and long-term economic growth of Canada at the political forefront. The current state of Canadian environmental protection is undergoing a rapid pace of change with declining funds, tighter incentive restrictions, and weakening environmental laws and policy amendments (e.g. Jobs, Growth, Longterm Prosperity Bill C-38). According to annual federal budget plans ‘Species at Risk’ funds have declined from 100 million/yr in 2007 (The Budget Plan, 2007. p. 73; www.budget.gc.ca/2007/pdf/bp2007e.pdf) to 25 million/yr in 2012 (Economic Action Plan, 2012. p. 138;

www.budget.gc.ca/2012/plan/pdf/Plan2012-eng.pdf). Environmental groups and scientists argue that the Canadian federal government has delayed putting species on the list, failed to determine delisting requirements, or develop and implement recovery plans to protect habitat and restore populations (e.g. Case study review, Dawe and Neis, 2012). A prominent examination is needed to assess shifting Canadian governance and how existing environmental policies will affect what is being observed. Addressing the direct and indirect environmental effects from agricultural and climate change policy, including failure to include climate change in planning needs to be explored. As well as, accepting a reality where the ‘cost is to high’ and the economic threshold in ecological protection and restoration lead to the inevitable acceptance of novel ecosystems. The south Okanagan Valley’s high biodiversity, agro-economy, and extreme climate may be a sensitive location for examination of such a socio-economic and ecological tipping-points.

Despite the present political environmental conditions, the need to maintain wildlife populations is an environmental priority shared among key governing agencies in Canada (Federal-Provincial-Territorial Task Force on the Importance of Nature to Canadians, 2000). The first global agreement on the conservation and sustainable use of biological diversity in Canada was signed in 1992 at the Convention on Biological Diversity (CBD) (United Nations Environment Program, 1992). From the CBD agreement emerged a need to understand the effects of human activities on biodiversity (Federal-Provincial-Territorial Biodiversity Working Group, Biodiversity Convention Office, Canadian Biodiversity Strategy, 1995).

Another supporting agricultural policy in Canada, The Agriculture Policy Framework (APF), focuses on farming using environmentally sustainable practices (Agriculture and Agri-Food Canada, 2002). Canadians have the opportunity to contribute to biodiversity and sustainability of the land through stewardship. Non-government organizations utilize funding such as the Federal Habitat Stewardship Program to promote grassroot community and private landowner stewardship. Agriculturalists are in a position to make significant contributions as land stewards to the protection of habitats, species, and sustainability of their sector (Lütz and Bastian, 2002). The APF provides farmer’s opportunities to develop site specific Environmental Farm Plans (EFP) and the eligibility to apply for financial and technical assistance through the National Farm Stewardship Program (NFSP).

To address the diverse agricultural landscapes across Canada the NFSP has been designed to allow provincial flexibility to support Best Management Practices (BMP) that address regional biodiversity priorities. Agricultural land-use and wildlife habitat information can best be gathered regionally and locally, where planners have the opportunity to work with landowners to set habitat goals and objectives that meet the needs of a variety of wildlife species. For the most part farmers understand the value of conserving wildlife and natural habitat elements, but extension practice and incentive programs can further this understanding to enable stewardship action. A calculation of variable land margins proposed for habitat improvements found removing six percent of agricultural land from cultivation can occur without negative financial effect for farmers (Lütz and Bastian, 2002). Initiatives to encourage the adoption of BMP are underway across Canada and supported by various governmental agencies and environmental organizations. Environmental Farm Plan conservation actions may incorporate BMP such as conserving riparian areas and natural habitat elements, expanding buffer strips, adopting conservation tillage systems, implementing rotational grazing systems, or adopting integrated pest management systems (Wind, 2003). The Canadian and Provincial governments' commitment to agricultural biodiversity and sustainable farming practices is being put into practice using a voluntary 'bottom-up' participatory approach to agricultural conservation planning. To meet Canadian biodiversity and sustainable agriculture goals at a national scale the understanding of land-use change and associated outcomes on habitat availability to support biodiversity is needed. Understanding the pressures on land and the opportunities available to support biodiversity is paramount and poses considerable challenge.

The National Agri-environmental Health Analysis and Reporting Program, initiated by Agriculture and Agri- Food Canada in 1993, constructed a set of environmental and Wildlife Habitat Indicators (WHI) to assess environmental conditions, risks, and changes resulting from agricultural actions (McRae et al. 2000 as cited in Javorek et al., 2007). The WHI assesses change in agricultural land-use and concurrent changes in wildlife habitat on Canadian farmland. Global economic pressures and human population growth principally drive agricultural land-use. The greatest concerns to conservation planners are land-use actions that affect the capacity of the land mosaic to support species. The occurrence of a species depends not only on the site characteristics but also in the context of surrounding landscape, such as degree of isolation (Bennett et al., 2006). Deterministic changes in land transformation lead to fragmentation, degradation, non-native species invasion, chemical misuse, expansion of human settlement, and road systems (*see* review Bennett et al., 2006).

Transportation infrastructure can be used as an example to illustrate the profound impact of deterministic change on the landscape. Highways and roadways are not only important features of the human constructed landscape, but are of particular consequence to wildlife especially with regards to species movements (*see* review Dodd et al., 2004). There are many complex structural features of transportation infrastructures (e.g. raised road bed, density) and characteristics of road-use (e.g. speed and volume). Understanding wildlife behavioral traits around road-use is an emerging field (Ciuti et al., 2012; Jaeger et al., 2005). The road system effectively serves as facilitators and barriers to species dispersal, habitat fragmentation, and results in significant species mortality due to vehicle collisions (Hels and Buchwald, 2001). Expansion of the transportation infrastructure is reflected in the irreversible pace of land development, and reinforces the need for a quantitative approach to assess habitat and change.

The current version of the WHI assessed changes in habitat availability of 493 terrestrial vertebrate species (birds, mammals, reptiles, and amphibians) on Canadian agricultural land over two decades (Javorek et al., 2007). A 5% decrease in habitat capacity occurred on Canadian agricultural land during 1981 to 2001. The habitat loss was associated with an expansion in cropland and a corresponding decline in pasture (Javorek et al., 2007). Due to direct structural and functional change to landscape processes the connectivity or fragmentation has the ability to alter wildlife populations in response to habitat change. The WHI does not consider habitat quality or adequately address the ecosystems services provided by wildlife or habitat. Natural and restored habitats that support species can provide significant benefits to agriculturalists, such as water management or pollinators.

An innovative way to examine habitat quality and effects of agricultural practices on wildlife is through 'Scenario based alternative future' modeling. The scenario-model examines habitat quality by using local scale information, while considering implementation of alternatives (Mörtberg et al., 2007; Santelmann et al., 2006). Advances in landscape ecology, geographic information systems, and computer modeling of ecological and economic processes provide planners and policy makers with tools that engage people visually. Moreover, to adequately inform policy, indicator and monitoring programs need to address analyses at a scale appropriate to detect effects of land-use change. A comprehensive approach to assess habitat change incorporates emerging anthropogenic threats relevant to agricultural land-use. Policies and programs designed to sustain biodiversity should not be developed independently of socioeconomic factors or policies favouring an increase in agricultural output per hectare, known as agricultural intensification (Benson and Willis, 1988). A holistic approach to policy-making decisions may be most appropriate to environmental and economic sustainability in the Canadian agricultural landscape

It is unlikely that the Canadian Federal, Provincial, or Regional governments will change policies to ensure the preservation of natural habitat elements as a means to support biodiversity on agricultural land. This may be in part because the Canadian agricultural lobby group influence is the most pervasive force to effect the economy, legislation, and in turn biodiversity protection. Improving upland terrestrial habitat for Okanagan amphibian species, and other nontarget taxa, will need to be approached within the current framework of voluntary stewardship. The future direction of the South Okanagan Wetland Restoration (SOWR) project will strategically target agriculturalists surrounding wetlands. Stewardship will be the mechanism to encourage protection and enhancement of native sandy soils, and effort will be made to educate farmers on less disruptive agricultural practices. The use of geographic information services and modeling programs, such as 'scenario based alternative futures' (Santelmann et al., 2006), will be used to examine the effects of compaction and agricultural practices on wildlife habitat. Regardless of the success of the SOWR project, policy makers need to be aware that small-scale conservation efforts to save biodiversity on agricultural land will be futile.

2.1.1 Agricultural transitional zones

The practical implementation of agricultural landscape plans should not be considered in isolation of surrounding land-use planning. Ecological, planning, and economic literature underscores negative associations among urbanization and agriculture to natural ecosystems (Zhang et al., 2008; McKinney 2002; *see* review Crossman et al., 2007; Soule 1991). Landscapes often straddle a transitional zone where urban and rural anthropogenic features interact. The rural/urban interface, or peri-urban zone, is a critical area of land cover change, leading to transformations in the ecological and geo-morphological systems and cause edge effect disturbances to wildlife. Rapid population growth in the rural/urban interface leads to the conversion of agricultural and open spaces into high intensity developed land uses that ultimately damage the land (Theobald, 2005). For the planning field, the best way to maintain wildlife and ecosystem values is to minimize habitat fragmentation and link habitat elements by optimizing corridors and linkages between the configuration and arrangement of habitat patches (Roe and Georges, 2007; Soule, 1991). The perpetual human land-use pressure within the rural/urban interface consequently results in an ever-reducing patchy mosaic of remnant habitats (Crossman et al., 2007). The international ecological literature provides substantial evidence that sustaining biodiversity among remnant habitat patches is unlikely (McKinney, 2002; Lehtinen et al., 1999; Barrett and Guyer, 2008, Amphibians). Yet, examples supporting the protection of small habitat patches within transitional zones do exist (Cornelis and Hermy, 2004; Sorace, 2001; Soule, 1991; Barrett and Guyer, 2008, Reptiles). In Italy, urban agricultural parks are thought to strongly improve opportunities to wildlife by adopting less intensive agricultural practices (Sorace, 2001). In Belgium, some small park habitat patches are considered 'hotspots' of biodiversity along rural/urban fringes, particularly if the park consists of a diverse number of semi-natural habitat types (Cornelis and Hermy, 2004). Small habitat patch preservation can become stepping-stones that enable species movement through a landscape, particularly where natural habitat elements are lacking. Countrywide protection of hedgerows in the United Kingdom connects an ecological network of landscapes, establishing a president of countrywide policy to protect corridors (summary by McCollin, 2000). There is a need in the Canadian agricultural context to consider the value of habitat corridors and the function of small isolated habitat patches for a diversity of species.

In Australia's agricultural regions small isolated protected areas are insufficient to conserve biodiversity (Crossman et al., 2007). Australia has identified the urgent requirement to restore habitat in agricultural and urban interfaces to halt species decline. In Australia there has been a move towards regional biodiversity planning and goal setting, however specific details on how to plan for achieving targets in complex landscapes is lacking (Crossman et al., 2007). Striking a balance between attaining maximal ecological benefit while having minimal economic impact is a common feature challenging both the Australian and Canadian governmental efforts aimed at biodiversity protection. The Australian conservation targets use a systematic landscape restoration model for a mixed-use rural/urban fringe landscape that produces sufficient solutions to meet comprehensive and adequate preservation goals (Crossman et al., 2007). Unlike the Canadian WHI model, the Australian model incorporates landscape connectivity and considers its influence on species dispersal. Australia is moving towards a biodiversity planning process that implements practical bottom-up planning blended with a top down approach, as seen in active adaptive planning management. The suggested approach for conserving Australia's biodiversity in the urban/rural interface is a transparent systematic approach inclusive of landowners and community level consultation. Similarly, Canada incorporates both a top down (policy level protection) and a bottom up (stewardship) approach, however seems to lack the incorporation of large-scale habitat connectivity as a means of biodiversity protection.

2.1.2 Implications

Conserving biodiversity and maintaining environmental integrity are essential to long-term human interest and global health. Governments play the primary responsibility to orchestrate efforts to ensure Canadian biodiversity. The Canadian and Provincial governments utilize a rational comprehensive top-down approach to conservation supported by policy. However, a Transactive bottom-up stewardship model enables practical environmental action. More than ever, achieving environmental sustainability in agriculture has become a pressing and complex challenge. Although converting natural habitats to agriculture has accrued short-term benefits to agricultural producers, the long- term societal benefits of keeping natural lands intact outweigh the short-term gains from degrading such habitats, e.g. soil erosion (Powers, 2010). The conservation of many species requires simultaneous management of multiple landscape features, such as natural habitat elements, corridors, and small remnant habitat patches. Planners working within rural, urban, and natural areas need to understand mechanisms underlying the responses of species to a mosaic structure and complex interfaces. The consequences of landscape change, for a range of ecological processes, will offer valuable new insights to policy planners that will filter down to enable agriculturalists to make well informed choices when it comes to land use. Experience gained from professional practice demonstrates implementation of regional biodiversity goals on agricultural lands within the current provincial and federal framework. Yet the Canadian economic framework does not directly address sustained biodiversity. Therefore, lessons learned from practices used in other countries should be explored. Until the driving force of economics is equally balanced with environmental integrity our loss of biodiversity on a global scale remains the greatest threat to humanity in the twenty-first century.

2.2 restoration ecology and ecological planning within degraded landscapes

Protected areas are critical for conserving the world's biota, but in a human-dominated landscape long-term persistence of many species will depend on their capacity to survive within the occupied area managed by humans. Landscapes are mosaics of natural, agricultural and urbanized land uses, interspersed with vegetation patches, roads, and various water elements. Such land mosaics offer an array of habitats for floral and faunal species. Aspects of the surrounding landscape influence the occurrence of some species, including proportion of land use, proximity to habitat elements, measures of isolation, and occurrence of connecting habitat (Bennett et al., 2006). Fragmentation of habitat into isolated patches can have profound implications on mobile species that require several adjacent habitats to sustain their life cycles (Lehtinen et al., 1999). The relative amount of modified habitat, particularly agricultural and anthropogenic development, around ponds has been used as a reliable indicator of salamander

> Box 2. **Ecological Structure and Function.** Despite goals to restore ecological function, we often assess measures of restoration structure for success (Grayson et al., 1999). Ecological function includes the interactions of organisms with one another and the physical environment. Ecological functions require examination of processes through time, such as persistence of a species, productivity, or recycling of nutrients. Structure measures what is present at one specified sampling time. Unfortunately the sampling designs to measure function over time are increasingly complex. Structure can often be assessed over a few years, whereas function may take decades.

genetic population isolation (Greenwald et al., 2009). At the landscape scale, the connectivity of habitat for species is critical for effective conservation and ecological planning. Understanding patterns of habitat selection and microhabitat conditions are important to provide appropriate management for species conservation (Block and Morrison, 1998). Species may be restricted to natural or semi natural habitat elements, while some species might use anthropogenic alternatives.

An ecosystem may be in various states of natural disturbance depending on the frequency and intensity of environmental conditions and disturbance regimes, hence selection pressures and other evolutionary factors like genetic drift will vary (Marty, 2005). As such, natural disturbance plays a critical role in maintaining the diversity, structure and function in many ecological systems (Box 1). Anthropogenic disturbances may exceed abilities of populations to recover and the outcome is a reduction in the overall structural and functional complexity, with a resultant decline in ecosystem capability (Grenfell et al., 2007). The purpose for restoring degraded landscapes is to enhance ecological processes and structures. Ultimate threats to biodiversity, including expanding human population and global markets for resources are generally beyond the scope of ecological planners. These ultimate threats operate from a broad social, economic and political origin. In contrast, ecological planners can address proximate threats that affect biodiversity on regional and local scales. Proximate threats include habitat conversion, urbanization and recreational use, natural resource harvesting, and the impact of invasive species (McKinney, 2002; Pressey et al., 2007). Restoration ecologists and ecological planners focus on the maintenance of ecological processes, while balancing conservation with rapid development. Habitat restoration aims at ensuring further habitat loss or degradation does not occur or is reduced by returning degraded patches to a more natural and/or functional state (Grayson et al., 1999).

Ecological planning is inherently spatial, interdisciplinary, and is emerging as a transdisciplinary (Naveh, 2005, 2007) field that addresses protected areas, nature reserves, rural landscapes, and urban planning. Managers have a tendency to work in isolation from the research world, with decisions being rarely based on scientific evidence, and with few available assessments of effective conservation action (Grayson et al., 1999; Armstrong and McCarthy, 2007; Pressley et al., 2007) possibly due to a business model versus academic or governmental approach. The high rate of failure among wetland restoration projects, using a single site approach, has warranted the integration of a watershed or landscape scale approach integrating science and management. Feasibility studies evaluate whether a restoration effort should be attempted and identify the potential downside and knowledge gaps. Stakeholders discuss and contribute to project design in advance to resolve any conflicts. Emerging adaptive management strategies, feasibility frameworks, and decision support systems have increased project success when establishing restoration feasibility by integrating science and management into one framework (Steyer and Llewellyn, 2000; Euliss et al., 2004; Hopfensperger et al., 2006, 2007; Davidson and Finlayson, 2007; Goosen et al., 2007). Integrating case studies in the decision-making process focus efforts on the ecological, social, and economic aspects of potential restoration (Hopfensperger et al., 2006).

Inclusive of the adaptive management strategy is conducting wetland resource inventory, assessing resource condition over time and contributing factors of degradation prior to restoration efforts, as well as prioritizing watersheds or sites, and determining restoration potential. It is important to understand the types of perturbations affecting wetlands before taking appropriate restoration action. Some disturbances and their corresponding responses may be sustained, short term, or a combination of these two (Grayson et al., 1999). An adaptive strategy recognizes the importance of remaining patches and understanding the regional heritage and human interactions. Implementation of the project using repeatable experiments, standardized monitoring, and systematic evaluation follows next. The use of consistent monitoring protocols for assigning condition, designing and evaluating restoration is of central importance (Brooks and Wardrop, 2006). Without evaluation, valuable resources potentially continue to be wasted, wetland habitats remain degraded, and methods are not assessed for future use. Poor assessment is the outcome of poorly defined non-predictive hypotheses, unrealistic goals, or inappropriate sampling designs to adequately assess the project (Grayson et al., 1999). In practice, evaluation of restoration projects is very low. In the United States, only 10% of more than 37,000 river restoration projects had documentation and monitoring protocols in place (Bernhardt et al., 2005). Implementation of large-scale restoration is facilitated when adaptive management principles are embedded throughout the organizational structure of legislation (Steyer and Llewellyn, 2000).

2.2.1. Ecological planning for wetlands

Wetlands play an essential and significant ecological role in their diverse structural and functional characteristics, covering only 6% of the earth's surface (Mitsch, 2005). Many of today's global environmental issues can be significantly tied to the function of wetlands. There is a functional interdependency of wetlands, which together operate as complexes in watersheds; their functional losses are also felt collectively. Wetland functions influence water quality, air pollution, global warming, soil erosion, flood and drought control, and also the decline of wildlife populations, and the degradation of habitats. Human modifications of terrestrial and aquatic habitats can have an accumulative effect on many large-scale landscape processes (*see* review Norberg, 1999; Qian and Linfei, 2012).

Throughout history, people have altered wetlands dramatically through the mistreatment and manipulation of the land. The conversion of wetlands to agriculturally productive lands has occurred worldwide over centuries, and accounts for disappearance of 70% of freshwater wetlands (historical review, *see* Biebighauser, 2007). Less than 50% of the world's wetlands remain today (Mitsch, 2005). Despite the functional role of wetlands, continued deterioration occurs at a rate greater than any other ecosystem globally (Davidson and Finlayson, 2007). Europe and New Zealand have the greatest amount of wetlands lost (> 90% destroyed), followed by North America, China, and Australia with greater than 50% of wetland lost (Mitsch, 2005). An imbalance exists between the scale at which wetland losses are accepted and the scale at which wetlands are preserved. Globally, a few highly significant large-scale restorations of wetland systems are emerging. An ecological engineering approach is being used to reverse historical human engineering feats that transformed landscapes (Mitsch, 2005). Denmark's largest river, the channelized Skjern River, is being returned to a meandering state with an estimated cost of $254 million. The restoration and retransformation of the Florida everglades to a more natural state is estimated at $8 billion dollars. Additional large-scale restoration examples include Delaware Bay, USA; the Mississippi River Delta, USA; Louisiana costal area, USA; and the Mesopotamian marshlands of Iraq (for examples and review, *see* Mitsch, 2005).

Efficient conservation initiatives need to be undertaken at an appropriate spatially explicit landscape level, incorporating both structural and functional processes that interact with a mosaic of multiple-use habitats where people and wildlife cohabit. Conservation efforts, particularly in human occupied areas, should be designed to actively involve groups or people living within or near the restoration site efforts. Transactive planning will be valuable to local restoration efforts, particularly where there is a lack of governmental capacity or will towards conservation action. Researchers, practitioners, management agencies, landowners, and the public relate to degraded landscapes from different perspectives, but together can contribute to ecologically meaningful solutions (Grenfell et al., 2007). The field of ecological and conservation planning is currently going through a paradigm shift towards a transdisciplinary approach. A network of managers and natural scientists integrate geophysical and ecological aspects, bridging with the natural and cultural values of human interactions (Naveh, 2007). Even if an organization or stakeholder's focus is narrow, their activities need to be recognized as a potential source of biodiversity and ecosystem stewardship. Ecosystem management is enhanced through restoration projects that target collaboration with multiple organizations and stakeholders, as opposed to groups or individuals working in isolation (Schultz et al., 2007).

A significant weakness in restoration ecology is the preponderance of uncoordinated local, micro-scale efforts, tied likely to funding of individual properties or projects. Typically, failed conservation programs unsuccessfully modify people's actions or lose the interest and support of volunteers. Behaviour-based principles of social marketing (Wilbur, 2006) and citizen-science (Oscarson and Calhoun, 2007) can be implemented into programs and successfully influence peoples' actions. The failure to consider the interests of local communities can result in a lack of support for conservation efforts and in subsequent degradation of the habitat (Ancrenaz et al., 2007). Inclusive community-based restoration projects build support through training sessions, community presentations, and environmental education. To illustrate, volunteer monitoring training programs, such as Frogwatch (http://www.env.gov.bc.ca/wld/frogwatch/) targets citizen-science as a valuable tool in collecting meaningful data to aid conservation planning. Community-based programs require long-term commitment to establish an accessible physical presence on the ground. Connecting with community life, respecting local traditions, understanding historic land-use, and valuing the knowledge and the connection people have with their land are essential to bridge existing gaps and build trust. Recognition of the intrinsic value of species that inhabit an area and the realization by community members that preserving the environment benefits their own well-being is critical (Tyler et al., 2007).

2.2.2 Amphibians as indictors of ecosystem degradation

Amphibian fauna can be indicators of the impact of anthropogenic disturbance to wetland ecosystems and the state of degradation in the landscape. This diverse group of species is widely distributed, occupies nearly all habitat types, and has a critical role in trophic interactions of population and in community ecology. Disturbances physically alter amphibian aquatic and terrestrial environment both directly and indirectly and, although cryptic by sight, frog species are easily detected by their call (Shirose et al., 1997; Nyström et al., 2002). Consequently, the susceptibility and response of some amphibians to perturbation are measurable as well as subsequent recovery. Using an indicator species, reinforces the importance of standard protocols and methodologies is essential to increasing compatibility when collaborating or comparing research findings. Incorporation of amphibians into the ecosystem assessment should be considered during the pre- and post-evaluation of wetland restoration activities.

To more effectively measure impact of restoration activities, the goal should be to monitor changes in amphibian communities both spatially and temporally from a local single species scale (e.g. population size, reproduction, and fitness) to a multi-species ecosystem scale (e.g.

hydroperiod, population modeling, and habitat management). Development and incorporation of habitat, population, and environmental modeling is an important tool for predicting amphibian responses to both continued degradation or for restoration measures (Compton, et al., 2007; Knapp, et al., 2003). The importance of continuing long-term amphibian studies spanning multi-generations is paramount to increasing data reliability and understanding changes due to restoration.

The ecological scientific community has devoted great attention over the past decade to the question of global amphibian decline and its relationship with ecosystem degradation. Scientists worldwide have documented declines in amphibian populations with no single stressor identified (Alford and Richards, 1999) and studies examining multi-stressor environments are rare (pathogen, contaminant, plankton, mesocosm study on species assemblages *see* Buck et al., 2011). Determining contributing factors to amphibian decline is fundamental to the establishment of recovery and global scale conservation actions. An expansive number of hypotheses contributing to population declines have been proposed and explored in laboratory and field settings. Ecosystem degradation and stressors related to amphibian decline include, but are not limited to, direct and indirect effects of habitat destruction, environmental contaminants, and climate change (Hecnar and M’Closkey, 1998; review Sparling et al., 2000; Alford et al., 2001). Biological systems rarely experience a single disturbance, which makes the comprehension of causal relationships and the cumulative and synergistic interaction effects of stressors increasingly complicated. It is important to note that amphibians globally represent one of the most understudied taxa (Sparling et al., 2000) and receives significantly less research funds compared to other vertebrate classes (Gratwicke et al., 2012). The alarming rate of global deterioration causes an immediate threat to human existence, as environmental indicator species amphibian research has been launched into a rapidly new era of discovery.

2.2.3 Amphibians as indicators of poor habitat

The vulnerability of amphibians to disturbances on land, in the water, and in the atmosphere makes them particularly good indicators of ecosystem health. Smooth permeable skin, a complex life cycle involving metamorphosis, and dependency on both aquatic and terrestrial habitats increase amphibian susceptibility to environmental changes. The contribution of direct and indirect effects of habitat destruction to reduced amphibian abundance, diversity, richness, and distribution is universally acknowledged (review Hecnar and M’Closkey, 1996, 1998; Gibbs, 1998). Urban and agricultural development has resulted in the expansive clearing of terrestrial land (Hecnar and M’Closkey, 1998) and in the substantial loss of wetlands due to infilling and dredging (Nyström et al., 2007). Terrestrial and wetland patches that escaped destruction often become isolated and fragmented through the landscape. Fragmentation affects and alters migratory movements, distributions, and corridors required to sustain metapopulation dynamics in amphibian populations (Gibbs, 1998; Soule, 1987 cited in Hecnar and M'Closkey, 1996). Fragmentation furthermore increases predation risk, energy resource consumption, and roadway encounter. Vehicular mortality is a significant demographic force, particularly for migrating breeding amphibian adults and dispersing juveniles, and has been implicated in limiting populations on a regional scale (Gibbs and Steen, 2005; Nyström et al., 2007). Road networks have expanded dramatically over the last century and now affects a significant portion of the land (Gibbs and Steen, 2005). Species respond to roadways and traffic intensity in various ways, but as few as five to 26 vehicles/hour significantly increases mortality for some species (Mazerolle, 2004). A comprehensive study looking at increased traffic intensity correlated the number of calling males frogs and the disappearance of populations in Sweden (Nyström et al., 2007). Dispersal of fossorial species unable to penetrate cropland landscapes might result in isolation around natal ponds (Gray et al., 2004). Dispersal mortality is particularly high when migrating amphibians cross roadways (Gibbs, 1998; Hels and Buchwald, 2001; Hels and Nachman, 2002; Mazerolle, 2004). Solutions employed by large mammal conservationists to reduce road mortality have been attempted at the amphibian level including construction of fence lines and eco-passage tunnels under roadways (Lesbarréres et al., 2004). There are few published studies measuring or evaluating the effects of fences or tunnels on amphibian and wildlife movements, with most studies focusing on use (van der Grift et al., 2013).

2.2.4. Environmental contaminants: Amphibians as 'canaries in the coalmine'

As amphibians move through the landscape during their aquatic and terrestrial life stages many encounter air borne, water borne, and ingested environmental contaminant exposure. The number and types of contaminants, their sources, additive and interaction effects, and routes of exposure are exhaustive, as is the scientific literature on this topic (Carson, 1962; Bishop et al., 1997, 1999; Harris et al., 1998; Alford and Richards, 1999; Alford et al., 2001; Nyström et al., 2007). Environmental contaminants have been implicated in reduced species diversity and density, increased egg and embryo mortality, increased morphological deformities, reduced fitness, altered sex hormones and sexual traits, as well as genetic degradation, increased rates of predation, reduced motility, altered feeding behaviour and food web structure, increased susceptibility to disease and parasitic infection in amphibian species (*see* review Sparling et al., 2000).

2.2.5 Climate change

Similar to the pervasive exposure of contaminants, changes in weather patterns are influencing amphibian life cycle patterns (Blaustein et al., 2001), aquatic habitat structure (Hoyle and James, 2005; Pyke and Marty, 2005), and rates of disease transmission (Blaustein and Dobson, 2006; Pounds et al., 2006;) around the world. Fluctuations in seasonal temperature and precipitation are implicated in altered breeding patterns (Blaustein et al., 2001), with implications on population structure. Seasonal weather influences amphibian behaviour and breeding onset. Prematurely deposited amphibian eggs may increase the risk of frost damage and late deposited eggs may be susceptible to incomplete development. Similarly, weather trends can alter the rate of tadpole development and metamorphic emergence, and can lead to early adult spring emergence or delayed hibernation, resulting in reduced glycogen levels of hibernating species with far-reaching impact on the overall fitness of the species (Storey, 2007). Increased fluctuation of water levels changes the quality and quantity of aquatic habitat. Surface area of wetlands, especially when connected to large water bodies, has been observed and predicted to increase (Hoyle and James, 2005). Conversely, drought conditions may lead to reduced hydroperiods and number of ephemeral habitats (McMenamin, 2008). As such, species requiring specific aquatic habitat features may not be able to cope with unpredictable changes in their habitat, possibly resulting in seasonal reproductive failure. Perhaps the most startling link between climate change and amphibian declines is that the onset of warmer temperatures has enhanced the rates of disease transmission of an often fatal fungus (Chytridiomycetes, Rhizophydiales, *Batrachochytrium dendrobatidis;* Longcore et al., 1999) among amphibians (Blaustein and Dobson, 2006; Pounds et al., 2006) and altered community structure (Searle et al., 2011). The rapid decline or extinction of many amphibian species has resulted in a global effort to address disease prevention, detection, and conservation efforts.

2.2.6 Wetland habitat restoration

Conservation of wetland habitats for amphibians has been largely ineffective at the federal, provincial, and state levels of government in North America (review, Oscarson and Calhoun 2007). To protect wetlands and wetland species, legislation in Canada focuses on the Water Act, the Fisheries Act, the Wildlife Act, and the Species at Risk Act. The most explicit statement 'no net loss' is the Canadian federal government's position on wetlands under the Federal Policy on Wetlands Conservation (http://publications.gc.ca/pub?id=9.686114&sl=0). Since this is a policy, rather than a law, it is not legally enforceable in court. Wetlands without water permanency are in a precarious position, being offered little protection. In contrast, the United States Wetland Protection Act applies to all wetlands types. In addition, statewide programs exist to prevent loss of wetlands during development proposals. The Natural Heritage & Endangered Species Program in Massachusetts has certified over 2000 ephemeral wetlands (Massachusetts Certification Guidelines, 2000). This progressive public program certifies ephemeral sites based on a photograph documenting the presence of seasonal water and evidence of breeding activity from any one of seven identified obligate ephemeral species, including fairy shrimp (Crustacea, Branchiopoda), salamander species (Caudata), and the Eastern Spadefoot (Anura, *Scaphiopus holbrooki*).

Some jurisdictions regulate or impose restrictions on wetland development, but few consider amphibian terrestrial habitat. In many regions wetland destruction is permitted if other wetlands are restored or constructed as replacement (France, 2003; Biebighauser, 2007). Wetland replacement strategies may take decades to reach the preexisting level of complex function and structure. The time suggested before wetland restoration can be judged is 15 to 20 years (Mitsch and Wilson, 1996, cited in Nedland et al., 2007), with some projects seeing measurable results after eight years (Pechmann et al., 2001) and 13 years (Petranka et al., 2007). Over the past 30 years the US Fish and Wildlife Service restored hundreds of wetlands. A reevaluation of a subset of Wisconsin sites noted that the use of the wetlands by ducks and anurans did not exhibit changes over a 12-year period. Although vegetation rapidly colonized restored sites, plant communities in the restored sites poorly resembled local natural wetlands plant communities. Loss in plant diversity and species richness over the 12 years was attributed to annual plants being replaced by perennial plants (Nedland et al., 2007).

2.2.7. Human developed landscapes

Agricultural and urbanized environments often provide marginal breeding habitat for amphibians and are generally assumed to have negative affects on amphibian populations (Gray and Smith, 2005); Rubbo and Kiesecker, 2005). Agricultural drainage ditches, retention ponds, and remnant wetland patches, in some cases may support some resilient amphibian breeding. Increased sedimentation, draining, dredging, and removal of aquatic vegetation occur in agricultural wetlands with high frequency. In some agricultural regions, natural and even unnatural wetlands are scarce.

The construction of artificial ponds represents an important alternative-breeding habitat for many species (Vasconcelos and Calhoun, 2006). Properly managed, these artificial ponds may effectively increase the total amount of breeding habitat and help sustain amphibian populations (Gibbons et al., 1997) by increasing species richness and abundance (Brown et al., 2012). Restoration of the local hydrology is the most important factor when considering restoration. Former wetlands that have been drained or which have had their water supply disrupted are most likely to be successfully restored if the factors that resulted in the loss of the wet ground can be reversed. A comparison of ten natural and thirty constructed ponds surrounded by livestock found that poor water quality (increased phosphates and turbidity) was correlated with reduced amphibian reproductive success (Knutson et al., 2004). Similarly, in small agricultural ponds species diversity, richness and reproductive success was closely associated with water quality (low nitrogen concentrations where cattle grazing absent) and less emergent vegetation characteristics (Knutson et al., 2004), unlike habitat variables in upland habitat. Water quality is a significant variable affecting constructed and natural pond success in agricultural landscapes. Agricultural management should consider the pond scale in addition to considering characteristics of the surrounding landscape (Knutson et al., 2004). Wetlands accessible by livestock mechanically disturb vegetation and crush adults, eggs and developing amphibian tadpoles. In addition to alterations in the water quality, livestock have been implicated in reduced surface area (Knutson et al., 2004) and decreased wetland hydroperiods. Agricultural habitats are commonly disturbed mechanically, resulting in maceration of amphibians and increased soil compaction that can influence burrowing behaviour in some species (*see* section 3.2.2 Terrestrial microhabitat).

Even though frequently disturbed urban wetlands may not easily support amphibian populations, they still provide recreational and educational functions to the community and, as such, their restoration is important (Grayson et al., 1999). Wetlands in urban areas are surrounded by significantly more disturbed upland habitat and far greater road density than rural wetlands. Small habitat remnants on private lands and public green-spaces provide limited refuge to wildlife threatened by urbanization. Amphibian populations in urbanized landscapes are significantly associated with reduced amphibian richness, water permanency, and predatory fish (Rubbo and Kiesecker, 2005; Knapp et al., 2007). Moreover, urban wetlands are often associated with environmental aesthetics and recreational use, such as golf courses (Hodgkison et al., 2007) leading urban planners to recommend small-scale conservation. Preserving a diversity of amphibian species in urbanized landscapes requires a variety of protected wetland sites that encompass various hydroperiods, adequate buffers, and are connected by dispersal routes, and lack fish (Rubbo and Kiesecker, 2005). Over the long-term, conservation of small fluctuating amphibian populations are likely to be unsuccessful.

2.2.8. Amphibian population dynamics

Wetland mosaics of sufficient density, diversity, and proximity play a role in maintaining amphibian population dynamics (Compton et al., 2007). Understanding natural variations in amphibian population size and breeding dynamics, in degraded and pristine landscapes, urgently warrant more long-term studies (Pechmann et al., 1991). Extrapolation of data from long-term amphibian studies aids in the complex understanding of population dynamics and have direct influence on conservation decision-making. Amphibian populations that are characteristically connected to each other by varying degrees of dispersal are referred to as metapopulations. The goal of many local restoration projects is the establishment of viable amphibian metapopulations. In isolated areas the construction of artificial wetland mosaics close to naturally populated amphibian populations provided evidence that artificial wetland complexes can restore population dynamics and increase population numbers (Pechmann et al., 2001; Petranka et al., 2007). Multiple and diverse wetland types serve as an insurance policy for metapopulations, particularly when asynchronous breeding dynamics occur among many local populations. Given the naturally high rates of annual reproductive failure among amphibians, the likelihood increases that in any given year at least one wetland will support a breeding population and sustain the metapopulation till the next year (Gibbs, 1998). Local extinctions are more likely in small ponds when less than ten adults frogs are detected (based on 160 ponds: Hecnar and M'Closkey, 1997a). Extinctions in amphibian metapopulations may occur frequently; recolonization of species permits persistence at the regional scale. Extinction risk increases significantly as the connectivity between wetlands decreases and amphibian dispersal mortality increases (Hels and Nachman, 2002).

Loss of connectivity between amphibian populations additionally has contributed to the introduction of predatory fish for human recreation into permanent water bodies and has resulted in altered amphibian assemblages on a geographic scale (Hecnar and M'Closkey, 1997a,b). Non-predatory native fish can impact amphibian populations by indirectly agitating vegetation causing physical disturbance to amphibian eggs and reduced hatching success. Predatory fish represent one of the most prominent predator to amphibians in permanent water bodies by directly consuming eggs and/or larvae. Some amphibian species when selecting breeding habitats demonstrate a predatory avoidance to sites with fish or other predatory species (Petranka and Holbrook, 2006). In the presence of fish, amphibians display smaller body size and reduced species abundance (McGarvie Hirner and Cox, 2007). Amphibian species not adapted to coexist with fish are likely to become sink populations and are prone to local extinctions (BeeBee, 1997; Knutson et al., 2004). Large bodied and high fecundity amphibians with plenty of refugia are more likely to survive increased predation pressure from native species of fish (Hecnar and M'Closkey, 1997b). Predation pressures imposed by non-native invasive frog species in western North America are analogous to the effects posed by predatory fish. The American Bullfrog (Anura, Ranidae, *Lithobates catesbeiana (Shaw 1802*) and, to a lesser extent, the Green Frog (Anura, Ranidae, *Rana clamitans*, Latreillie 1801) have been introduced throughout the world for human consumption frog farms, as fish bait, and through the pet trade. Unlike fish, the highly predacious American bullfrog can disperse easily across the landscape invading entire watersheds, which results in the mass loss of amphibian species richness (Ficetola et al., 2007).

2.2.9. Models for amphibian habitat conservation planning

Conservation of wetlands has generally focused on a small spatial scale, often limited to the boundary of the wetland with undersized terrestrial buffers (Hecnar and M'Closkey, 1997a; Compton et al., 2007). Small-scale management of amphibian habitats is an inadequate strategy, because many amphibians live most of their lives on land away from wetlands. While some amphibian species and individuals may have high site fidelity, others may use more than one wetland in a season and their upland habitat may overlap with multiple wetlands, which reinforces the importance of management and the maintenance of wetlands and their upland terrestrial habitat connectivity at a landscape scale (Compton et al., 2007). Broad scale efforts to address wetland connectivity are complicated by the number of wetlands in a region and by the difficulty in prioritizing wetlands (e.g. with high biodiversity) and their surrounding upland habitat (e.g. connected natural habitat elements). Probabilistic models have been used to predict amphibian site occupancy in a patchy landscape by estimating the required number of ponds and corresponding likelihood of species persistence (Knapp et al., 2003). More progressive are spatially explicit population models that use base-line scenarios predicting the consequence of future land-use alternatives (Mörtberg et al., 2007; Rustigian et al., 2003) and the probability of amphibian species persistence. The probabilistic and population models approach to amphibian species persistence is greatly complimented by long-term monitoring studies and the use of geographic information systems (GIS) systems.

The importance of terrestrial habitat when considering amphibian wetland management strategies has established vital recognition (Roe and Georges, 2007; Keyser et al., 2011, salamander sp.). Ideally, management and protection would reflect a heterogeneous group of wetlands together with terrestrial buffer zones (Roe and Georges, 2007) For large regions, amphibian niche modeling has been used to project potential geographic range occurrence, resulting in a probabilistic distribution area that prioritizes areas with maximum representation of all species in a minimal total area (Pawar et al., 2007). Niche modeling aids conservation managers in prioritizing regions where the greatest restoration potential should focus. A dominant wetland conservation-planning paradigm for amphibian species identifies a core habitat having a breeding site and extending circular terrestrial habitat zones. Cores habitat and terrestrial habitat zones are based on average migratory movements of species. Recent studies, based on species telemetry studies, recommend protection of core areas from 160 m to 386 m (Semlitsch et al.,2002; Semlitsch and Brodie, 2003) for wetland breeding amphibians. The core habitat management approach is easily transferable to policy and is suggested for less developed areas on a small single wetland and landscape planning scale. The core habitat model can be limiting in a complex rapidly developing landscape. Constraints of the model include the risk that too small a core is estimated and critical habitat elements may be omitted, or too large a core is estimated and limited funds are spent conserving nonessential habitat (Baldwin et al., 2006). In urbanized and developed landscapes, a spatially explicit habitat approach (Baldwin et al., 2006) considers the network of locally specific habitat elements of isolated and grouped breeding habitat.

Conservation planners, aided by GIS technology, identify, link, buffer, and protect discrete habitat elements within known migratory distances from breeding pools. A spatial strategy centers attention on critical amphibian habitat requirements for population persistence and targets land-use restrictions imposed on property owners in a cost effective manner (Baldwin et al., 2006). The conventional core habitat model focuses on conserving land in concentric rings around breeding pools. Realistically, today's complex landscapes require economic and ecological efficacy that is better suited to the spatially explicit habitat model.

2.2.10. Wildlife populations and ephemeral wetland hydrology

Cattail marshes, swamps, and bogs are familiar and easily identified wetlands with water permanency. More conspicuous are the non-permanent wetlands. Non-permanent wetlands are interchangeably referred to as ephemeral, vernal pools, or seasonal temporary ponds. For the purpose of this dissertation, I have selected the term "ephemeral" when referring to wetlands without water permanency.

Wetland classifications and biotic communities are largely determined by the frequency, duration, and the depth of soil saturation that leads to flooding (Brooks, 2004). While not fully understood, the long-term hydrology of small, isolated ephemeral wetlands is influenced by the physical properties of the site, as well as the size of the wetland, and its connection to local and regional ground water resources (Winter, 2001 cited in Brooks, 2004). Ephemeral wetlands generally require underlying substrates with slow permeability and typically have no direct inflow or outflow (Bauder, 2005). Wetlands require the relatively shallow depressions of a wet meadow. Ephemeral wetlands are more typically set in a landscape consisting of mounds and depressions with various shapes, depths, and connecting swales. Ephemeral wetlands acquire water from spring snowmelt and precipitation, with pool duration being most dependent on the total seasonal precipitation (Bauder, 2005). Water loss can occur through seepage, ground-water outflow, or surface-water outflow. Patterns in precipitation (Brooks, 2004) and evapo-transpiration represent the sum of evaporation and plant transpiration into the atmosphere (Boone et al., 2006). Ground-water outflow and surface-water outflow have the greatest impact on the rate of weekly water level change and the timing of ephemeral pool drying (Brooks, 2004). The degree of isolation and topographic position factors alone and in combination cause each wetland to respond in a characteristic manner to the amount and pattern of rainfall (Brooks, 2004).

In some climates, precipitation is confined to a distinct, low-temperature rainy season followed by a prolonged, high-temperature draught season (e.g. California, USA: Bauder, 2005). The timing of seasons is predictable; however climactic variability is unpredictable. Temperature and precipitation combined with the pattern and intensity between years contributes to the climatic variability (Bauder, 2005). Consequently, ephemeral wetlands may be increasingly sensitive to climatic shifts and their direct relationship between pool hydrology and productivity may be good indicators of global warming (Brooks, 2004). Under climate change predictions, ephemeral wetlands throughout northeastern region of North America will experience more episodic precipitation and increased evapo-transpiration, leaving ephemeral wetlands dry earlier in the year and remaining drier for longer periods (Brooks, 2004). In the future, drier hydrologic conditions are predicted for the northern U.S. prairies where high temperatures and increased evapo-transpiration are expected, despite the predicted increased precipitation (Poiani and Johnson, 1991, 1993b cited in Brooks, 2004, Neilsen et al., 2006). Climatic changes may cause early drought periods for wetlands later to be filled by seasonal downpours, but after larval amphibians have died. Focus on projecting changes in precipitation patterns is critical to assessment of climate change effects on isolated wetlands. However on a landscape scale hydro-period modeling of these highly variable environments has been found to be impractical due to the amount of detailed information required for individual sites (Brooks, 2004; Bauder, 2005), hydro-period response among wetlands is not consistent for modeling applications. Pools differing in landscape position have been found to respond differently to the same precipitation events. However, hydrology may be reasonably modeled for a set of local sites (Bauder, 2005) or individuals site models (Brooks, 2004) where high quality data are available.

Wetlands provide fundamental habitat-use to a plethora of floral and faunal species. In North America about half of all the waterfowl nest in permanent wetlands. Two-thirds of the commercial shellfish and sport fish are derived from permanent wetlands, making permanent wetlands economically and ecologically important wildlife habitats (Mitsch, 2005; Biebighauser, 2007). While numerous species benefit from the existence of ephemeral wetlands for foraging, and some are excluded such as fish species, only a relatively few species have evolved individual life history traits with specific adaptations to the variability and unpredictability of ephemeral habitats. Obligate or facultative wildlife populations using ephemeral habitat are limited to members of the invertebrate and amphibian communities.

Ephemeral wetland species can be either aquatic or semi-aquatic opportunists exploiting both permanent and ephemeral wetlands. Ephemeral specialists are adapted specifically to both wet and dry environments. Specialists place an emphasis on completing their reproductive cycle and may do so by desiccation, resistance or tolerance at one point in the life cycle. A partial non-dependent wet phase life stage is also supports a completed life cycle and includes the mechanism of amphibian metamorphosis. Ephemeral wetlands support a rich and diverse invertebrate community adapted to annual drying, the composition of the community is strongly affected by the wetland hydroperiod. Increased hydro-periods lead to increased invertebrate richness, but the relationship between the two is complex (Schneider 1999, cited in Brooks, 2004; Watts and Didham, 2006.). Subtle changes in environmental factors can have major implications for the long-term persistence of specialized wetland species. Annual climatic variation effects have shown to alter the micro-distribution and co-existence, as well as the presence and distribution, of some wetland species and result in the unpredictability of their population dynamics and microhabitats (Vignoli et al., 2007). Plant communities alter microhabitats through changes in plant variation and abundance. Pools with shorter hydro-periods have a greater abundance of trees, compared to a greater abundance and variation of annual and perennial forbs in pools with longer hydroperiod (Palik et al., 2007). As an example, increased precipitation resulting from weather phenomena, such as el Niño (1997/98), was found to increase the rate of invasive grasses into the California ephemeral wetlands and consequently altered the pool hydrology and microhabitats for native flora (Bauder, 2005). To persist in the changing ephemeral environment, both flora and fauna must cope with large annual and seasonal variations in the longest continuous periods of rising and falling water levels during the rainy season. The fluctuation in terrestrial and aquatic conditions significantly affects the ecological balance of wildlife populations.

2.2.11. Amphibian species and ephemeral habitats

Amphibians represent an extraordinary, diverse and evolutionarily unique taxonomic group compiled of more than 6000 species worldwide that exist in a wide-ranging type of habitats. For the most part, amphibian species are biphasic: they have an aquatic life phase for breeding, egg-laying, and larval development, followed by a terrestrial upland foraging during the wintering phase. Many individual amphibian species exploit a large variety of wetland habitats, and move across terrestrial habitats occupying different wetland habitats within a season and from one season to another. Some salamanders breed in permanent ponds, and then travel to temporary ponds characteristically rich in high-energy prey to feed (Denoel et al., 2007). Variable habitat-use by salamanders resulted in different feeding habits, energy intake, and higher fitness among individuals as a consequence of the habitat use. Wetland hydroperiod is critical for the annual production and fitness of pool-breading amphibians. Amphibian metamorphic success increases with pool duration (Denoel et al., 2007). Additionally, amphibian metamorphosis is delayed in longer hydroperiod pools and produces larger individuals, which affect the fitness and survival of the species (Denoel et al., 2007). Amphibian species that are facultative or preferred breeders in ephemeral wetlands include the Mole Salamanders (Ambystoma sp., 13 - 24 weeks, Brooks, 2004), Wood Frog, (Anura, Ranidae, *Rana sylvatica*, LeConte 1825*:* 8 - 19 weeks, Brooks, 2004), and species of the Pelobatidae Family (3 - 6 weeks, Klassen, 1998). In addition to these species, Massachusetts, U.S., in a progressive approach to protecting ephemeral habitats and their species composition, formally designates the Spotted Salamander (Caudata, Ambystomatidae, *Ambystoma maculatum*, Shaw 1802), Blue-spotted Salamander (Caudata, Ambystomatidae, *Ambystoma laterale*, Hallowell, 1856), Jefferson Salamander (Caudata, Ambystomatidae, *Ambystoma jeffersonianum*, Green 1827), Marbled Salamander (Caudata, Ambystomatidae, *Ambystoma opacum*, Gravenhorst, 1807), and Fairy Shrimp (Crustacea, Branchiopoda, *Eubranchipus bundyi*) as obligate species (Massachusetts Certification Guidelines, 2000). Typically ephemeral species have evolved explosive, precipitation-driven synchronous breeding patterns: short periods of larval development and long periods of terrestrial use (Hall et al., 2002; Eggert and Guyétant, 2003; Jakob et al., 2003; Greenberg and Tanner, 2004; Arendt, 2006; Gomez-Mestre and Buchholz, 2006). Amphibian recruitment is often episodic; with partial or full reproductive failure in most years, but is then compensated by large cohorts in favorable years (Marsh, 2001 cited in Brooks, 2004). Climatic change that reduces water resources and increases evapo-transpiration further reduces the occurrence of productive years. Consequently, reproductive failure increases for species reliant on isolate wetlands. The number of wetlands with metamorphic success decrease along with the distance between these successful sites increases, resulting in reduced ability of juvenile dispersal, affecting colonization, and thus metapopulation dynamics and the maintenance of genetic diversity (Brooks 2004).

Extreme arid environments, coupled with specialized species like the Great Basin spadefoot (Anura, Scaphiopodidae, *Spea intermontana*), provide a sensitive model to examine the proximate and ultimate environmental threats, and assess the recovery response of aquatic species to restoration actions. The highly terrestrial and fossorial Great Basin spadefoot is uniquely adapted to breed in ephemeral wetlands. However, the Great Basin spadefoot is becoming increasingly at risk of decline due to the accumulative human impacts of their terrestrial and aquatic habitats. Environmental and ecological scientific literature and the SOWR project identified two main threats to Great Basin spadefoot survival. First and foremost, intensive agricultural and urban development has resulted in habitat fragmentation and permanently altered substrates, isolating spadefoot populations and reducing their connectivity between habitat patches. Secondly, global warming poses an increased risk of wetland loss and increased population isolation. Despite Great Basin spadefoots being highly adapted to living under fluctuating hydrological conditions, the extreme influence of global warming on hydrological period and larval development may exceed the species survival capacity. Without restoration and conservation actions to address the effects of proximate and ultimate threats, the long-term survival of the Great Basin spadefoot in the lower south Okanagan Valley is unlikely.

2.2.12. Conservation planning in practice

There are many complex factors interacting and contributing to the decline of amphibians globally with habitat loss the greatest causative factor (Alford, et al., 2001). The south Okanagan Valley is identified as one of Canada's most endangered ecosystems (Iverson et al, 2008; Cannings, 2000; Bryn et al, 1994), supporting many uniquely adapted desert species found nowhere else in Canada. Mapping analyses comparing the aerial extent of historical (1800s and 1938) and remaining ecosystem areas (2003) revealed substantial ecosystem loss of the Okanagan Valley (Table 2.1). The combination of ecosystem loss, fragmentation and degradation has had substantial impacts on local biodiversity and on the ability of species to exist, evolve, or migrate to other areas (Lea, 2008). The Lea (2008) report emphasizes the uncertainty about the earlier existence of smaller wetlands or vernal pools in the landscape, as much of the gentle sloping grassland had already been converted to agricultural uses when the 1938 air photographs were taken. Similarly, shrub riparian areas of water birch and red-osier dogwood communities may have contained small areas of wetlands, shallow open water, cattail marshes, and possibly other wetland types that no longer exist.

Table 2.1. Arial analysis for ecosystem loss by types in hectares for the Okanagan and Similkameen Valley, British Columbia taken in 1800, 1938 and 2003 (reconstructed from Lea, 2008).

Ecosystem	1800 (ha)	1938 (ha)	2003 (ha)	Percent loss (%)
Dogwood riparian wetland	15,209	4,497	1,208	92
Okanagan River	212	212	15	93
Cattail marsh	432	378	264	40
Idaho fescue blue bunch wheatgrass grass steppe	4,360	3,229	1,335	70
Big sagebrush shrub steppe	12,458	10,402	8266	33
Antelope brush needle and threat grass shrub step	9,896	7,325	3,178	68
Gentle slope grassland and shrub step	41,881	26,651	16,461	61
Low elevation wetlands (including marsh shrub swamp, meadow, shallow open water)	17,786	6,890	2,965	84

The Lea (2008) mapping analyses provide habitat-based evidence that lowland populations of wildlife are becoming increasingly isolated and disconnected from their aquatic and terrestrial habitats and from their upper elevation populations. The primary goal of the SOWR project is to restore and enhance small wetlands and to evaluate amphibian-breeding success as a measure of species and habitat recovery. For the purpose of simplicity and brevity, my position and defense will focus on amphibian habitat loss, putting all other factors aside. Nevertheless, factors such as environmental contamination, non-native invasive species, compacted impermeable soils are important to acknowledge in the cumulative impact on the decline of amphibians.

Similar to the SOWR project, most conservation planners and ecologists focus on amphibian breeding habitat and neglect the essential use of upland terrestrial habitat (Compton, et al 2007; Baldwain et al., 2006). Amphibians require both aquatic and terrestrial habitat protection in order to ensure long-term species biodiversity. No conservation effort to support biodiversity can be achieved fully without ensuring all necessary habitat elements are protected. The need for species to move across different habitat features to obtain resources, on a daily or seasonal basis, or at different life stages, should be considered. Moreover, adjacent habitat patches may influence amphibian suitability and use of a particular patch. Looking at a land mosaic helps to identify properties of various habitat elements, and provides consideration to the number, type, size, and spatial arrangement of elements within a habitat (Rustigian, et al., 2003; Bennett et al., 2006).

Considering factors that comprise a habitat mosaic will allow isolation of habitats and potential corridors to be identified. Stewardship programs implemented at a small scale, such as the wetland restoration in the presented example (*see* Chapter 4), require relatively few participants to significantly increase the amount of available amphibian breeding habitat. Understanding the land mosaic and improving upland terrestrial habitat for amphibians throughout an agricultural landscape is not likely to be achieved without policy to protect the natural habitat elements of the a given ecosystem. Stewardship programs with farmers on a small scale are achievable; however large-scale protection of habitat elements is both impractical and logistically infeasible. Even if habitat elements where identified it is unlikely that policy would be developed to address a single species needs like a frog, or even the needs of a taxonomic group such as the amphibians. Policy could be developed if natural habitat elements are identified as critical for maintaining ecosystem integrity. Natural habitat elements of the south Okanagan Valley have been identified, and as such, justified protection under policy.

2.3. CONCLUSION

The second chapter examines restoration ecology and ecosystem degradation and how their principles play an important role in the ecological planning and conservation of wetland species and their habitats. Science has aided us in defining the global extent of wetland loss, the vital function of wetlands and their connection with current environmental issues. Ecological planning can be aided by integrating environmental scientists, management, and the community into a transdisciplinary approach to restoration and conservation. The integration of strategic adaptive strategies, feasibility studies, targeted social marketing, and spatially explicit habitat models are progressive approaches being implemented into wetland restoration projects. Our recognition of the importance of wetlands, coupled with our increased sophistication in developing techniques for constructing and restoring aquatic ecosystems on the landscape, should give us a sense of optimism. Yet, wetlands and the species that inhabit them are encountering an ever-increasing level of threat and consequently are unlikely to be restored to a pre-degraded natural state. The Great Basin spadefoot model can be generalized to other amphibian species and ecosystems. The lessons learned from the amphibian species monitoring (Chapter 3) and the south Okanagan Valley restoration project (Chapter 4) can easily be applied into various restoration projects on a local, regional, and ecosystem scale.

3.0 SPECIES RICHNESS, DISTRIBUTION AND RELATIVE DENSITY OF ADULT AND EARLY LIFE STAGES OF AMPHIBIANS RELATIVE TO LAND-USE CHARACTERISTICS IN THE SOUTH OKANAGAN VALLEY, BRITISH COLUMBIA, CANADA (2003 TO 2006)

3.1 INTRODUCTION

The arid south Okanagan Valley of British Columbia is identified as one of Canada's most endangered ecosystems, and supports many desert-adapted species found nowhere else in Canada (Iverson et al., 2008). The combination of species being at their northern range limit, a restricted ecoregion, and highly human dominated landscape has contributed to the status assessment of local Okanagan species. Within the valley there are 71 species listed nationally as 'at risk' by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and a further 279 species listed provincially by the British Columbia and federal governments (Pearson and Healey, 2012). The valley supports the highest species richness in Canada (Warmen et al., 2004) and a strong economic community with a projected population of 102,000 by 2022 (South Okanagan Regional Growth Strategy, 2007). Consequently, much of the lower native valley ecosystems have been replaced by urban and agricultural development (e.g. soft fruit, vineyard), while livestock have intensively grazed the upper elevations of the valley. Since 1800, the Okanagan Valley there have been losses of 92% of dogwood riparian wetland, 84% of lowland wetlands, and 40% of cattail marsh ecosystem (Lea, 2008). A combination of habitat and ecosystem service losses, fragmentation and degradation has had substantial impacts on local biodiversity and on the ability of species to persist, evolve, or migrate to other areas (Lea, 2008).

There are many complex factors interacting and contributing to the decline of amphibians globally. Ecosystem degradation and stressors related to amphibian decline include, but are not limited to: direct and indirect effects of habitat destruction, environmental contaminants, non-native species, pathogens, and climate change (Stott et al., 1998; Sparling et al., 2000; Alford et al., 2001; Pounds et al., 2006; *see* review Blaustien et al., 2011). Ecological systems rarely experience a single disturbance, which makes the comprehension of the cumulative and synergistic interaction effects of stressors complicated. Habitat loss is the greatest known causative factor effecting amphibians globally (Alford et al., 2001; Gallant et al., 2007) and both wetland and terrestrial habitat losses in the Okanagan Valley are well documented (Lea, 2008) and pose an ongoing threat. Additionally, low egg survival in agricultural wetlands (Bishop et al., 2010), non-native invasive species (Lukey et al., 2012), road mortality (Crosby 2014), and unexplained die-offs of amphibians thought to be a result of acute environmental contamination (Ashpole et al., 2011) contribute to known local and potentially cumulative threats. The south Okanagan landscape now offers few protected areas, with the vast majority of wetlands located on private property. Assessing the availability of wetland habitat and land-use is critical in predicting which environmental variables influence species presence and distribution (*see* review Trumbo et al., 2012; examples include: invasive non-native species/elevation [Johnson et al., 2011]; roads/pond depth [Dai and Wang, 2011]; substrate/fitness [Janin et al., 2012]). Amphibian and turtle species' distributions may be influenced by many processes operating at various temporal and spatial scales; here we examine site-specific habitat data with broad scale human land-use categorization across a relatively small yet distinct geographical area in the south Okanagan Valley. Understanding species community structure and the factors influencing breeding habitat-use in a dynamic but degraded landscape are needed to direct stewardship and local conservation efforts. The goal is to identify wetland habitats and breeding pond suitability for amphibian and turtle species by assessing habitat-use and how landscape features are associated with the distribution of breeding populations to establish conservation priorities. We propose a hypothesis to explain amphibian assemblages in the south Okanagan Valley, B.C.:

Habitat Choice Hypothesis

Ha = If amphibian species richness, distribution and relative density are related to species-specific life history traits, then significant differences will exist among sites due to species specific habitat choices.

Ho = If amphibian species richness, distribution and relative density are not related to species specific life history traits, then no significant differences will exist among sites due to species specific habitat choices.

We report here the first, landscape scale wetland threats and amphibian and turtle occurrences in the south Okanagan Valley, B.C. This study quantifies species richness and relative density of early life stage of amphibian assemblages by repeat sampling of wetlands located in the south Okanagan Valley, B.C.

3.2 METHODS

3.2.1 Study sites

Wetlands studied were located in the south Okanagan Valley, B.C., Canada south of Penticton to the U.S. border in the tip of the long and narrow corridor of the Sonoran, Mojave, and Great Basin Deserts (roughly 70 km north-south and 10 km east-west) (Fig. 3.1). The number of and types of wetlands surveyed in each year varied due to property access, presence of water, and as a result of our detection of wetlands that we had not surveyed previously (N_{2003} = 24 sites; N_{2004} = 53 sites; N_{2005} = 71 sites; N_{2006} = 108 sites). Each discrete site was surveyed at least two times during the four-year survey period (2003 to 2006). Additionally, in 2006 calling amphibians were detected as we walked on the north side of the south Okanagan River channel dike and listened to calling from non-discrete ephemeral sites or permanent wetlands in the low areas immediately below the dike on the floodplain (N_{2006} = 164 records). Amphibian and turtle occurrences were also observed incidentally (i.e. occurrence data collected during a non-survey time period), including animals observed dead or alive on the road and photo-validated landowner sighting.

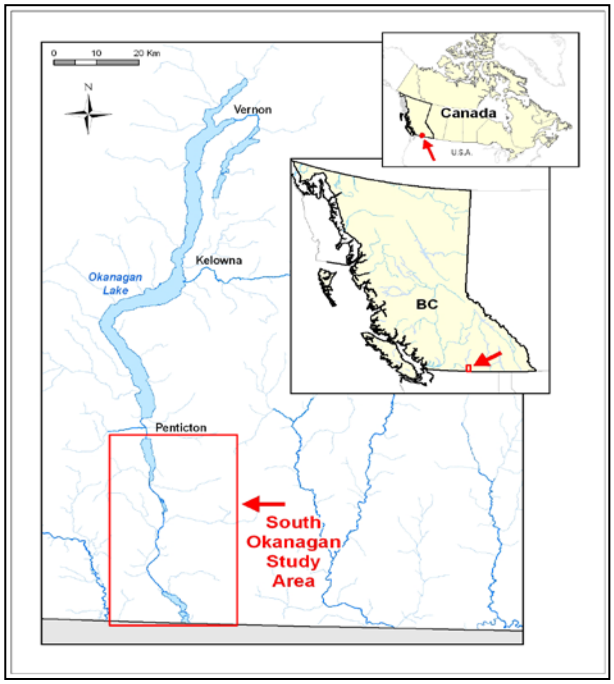

Figure 3.1. South Okanagan Valley study area, B.C.

3.2.2 Site classification and habitat parameters

All wetland sites and species localities were broadly classified as either within the lower valley floodplain (N = 96 discrete low elevation sites, elevation < 399 m) or in the surrounding upland valley benches (N = 33 discrete high elevation sites, elevation > 400 m up to 1135 m). All discrete wetland sites were classified according to water permanency (permanent vs. non-permanent) and the presence of fish (no fish detected, native fish species only detected, non-native fish species detected). At one site a change in classification occurred between years when the introduction of non-native fish into a previously vacant (no fish ever detected) wetland occurred.

Wetland sites were classified based on their dominant land-use (main-class) and then sub-classified according to land-use practices (sub-class) as follows: reference and non-grazed (N = 10 sites); agricultural (conventional pesticide/herbicide-use orchard N = 18 sites; certified organic orchard N = 14), livestock grazed (pond protected from livestock with fencing N = 10 sites; pond unprotected and livestock has water access N = 39 sites), and miscellaneous anthropomorphic sites (residential N = 7; artificial pools or ponds N = 4; ponds in golf courses N = 5). Additional categories included observations made on roads and roadsides (road classification: N_{total} = 130, highway N = 48 records, primary road N = 46 records, secondary road N = 36 records); and calling that was detected below the dike or heard in the distance while walking a transect along the dike trail parallel to the Okanagan River (channel classification: N = 164 records, Appendix 3.1). In four cases, land-use classification changed by year according to changes in land practices (e.g. unprotected grazed site became a protected grazed site with the addition of exclusion fencing). Not all main classe sites were represented in each year.

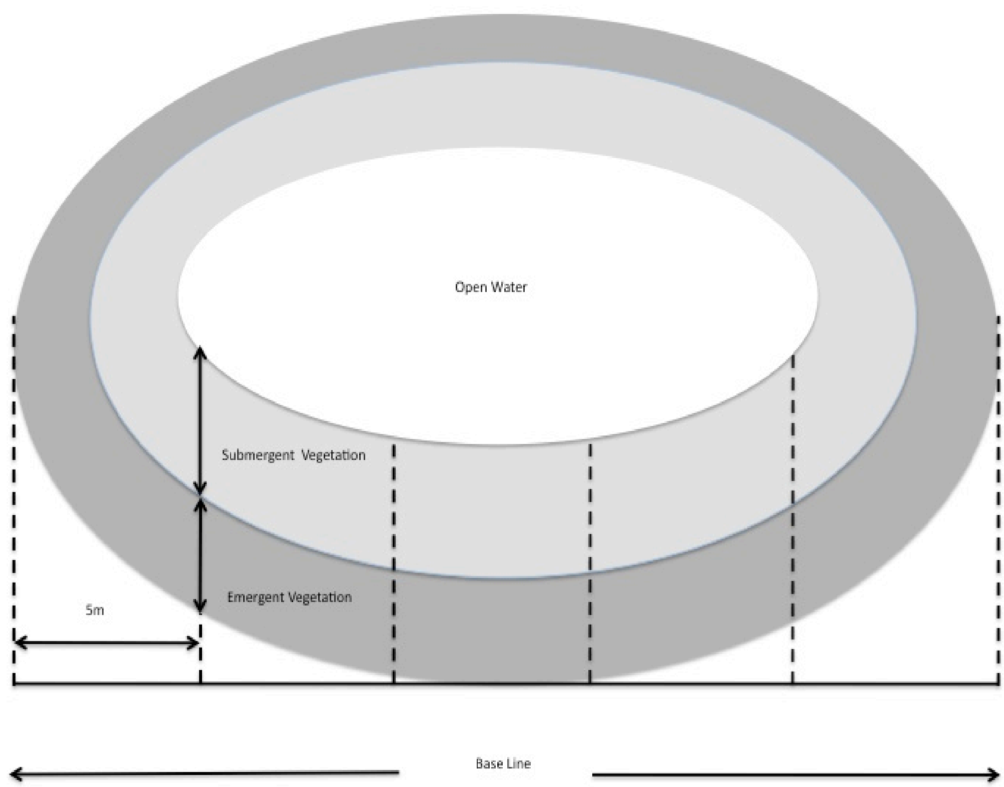

Figure 3.2. Pond schematic depicting transect survey design used to measure the emergent and submergent vegetation zones approximately every five meters (5 m) at a subsample of wetland sites (N = 39).

All sites (N = 108 sites) were assessed for anthropogenic stressors using frequency counts, including the presence or absence of: water withdrawal or discharge; infilling or shoreline modification; burn pile / intentional garbage dumping; introduced non-native invasive species (stocked fish / American bullfrog [*Lithobates catesbeiana*]); agricultural input (e.g. pesticides, herbicides); nutrient input (unrestricted livestock, turf fertilization); and artificially constructed sites. In 2003 and 2004, habitat characteristics were measured at a sub-sample of discrete pond sites (N = 39 sites), including: pond perimeter; maximum water depth; distance from the high water mark to agricultural crops; distance to nearest road; width of riparian and/or edge vegetation, emergent and sub-emergent vegetation was measured along a parallel baseline using perpendicular line transects at approximately five meter intervals throughout the length of the wetland (Fig. 3.2).

3.2.3 Water chemistry

Water samples were collected from wetland sites during late April to early May (early sampling: ovi-position period) (subset data from Bishop et al., 2010) and/or late June to early July (late sampling: metamorphosis period) annually. The number of sites and the number of samples collected per site varied annually, subject to the typical financial and logistical constraints (Table 3.1). Water samples were collected by hand into two 500 ml white semi-opaque plastic bottles cleaned with non-phosphate soap from approximately 5 cm below the water surface and approximately one meter from the shoreline. Water samples were collected, stored, and shipped the same-day before noon on ice (~4°C) to the Pacific Environmental Science Centre (PESC), Environment Canada in Vancouver, B.C. Samples for water chemistry were analyzed using standard analytical methods (PESC, 1999) for pH, conductivity, total nitrogen (TN), phosphorous o-PO_4 dissolved, total dissolved phosphorus (TDP), total phosphorous (TP), turbidity, chloride (Cl), fluoride (F), sulfate (SO_4), bromide (Br), nitrate (NO_3), nitrite (NO_2), phosphate (PO_4) and biological oxygen demand (BOD) (as per methods described in Bishop et al. 2010).

3.2.4 Study species

Study species of the south Okanagan Valley, B.C., included eight amphibians and one turtle species (Table 3.2). The locally extirpated Northern leopard frog (*Lithobates pipiens*) was designated by COSEWIC as Endangered in 2000 is currently restricted to one B.C. population in the Creston Valley (350 km west of the Okanagan Valley). Given this situation, we did not expect to detect this species. The invasive and non-native American bullfrog was first officially observed in 2003 in the Okanagan Valley. Given its threat posed to wetland species and their potential to become widespread, this species was monitored (*see* Lukey et al., 2012).

Table 3.1. Mean (standard deviation) water chemistry parameters collected from wetland monitoring sites (2003 to 2006: $N_{sites} = 50$; $N_{sample} = 152$) in early (late April to early May) and/or late spring (late June to early July), south Okanagan Valley, B.C. Differential Function Analysis was used to determine statistical differences among land-use sub-classes by pooling late spring samples (*****) over the period 2003 to 2006. Similarly, to determine statistical differences between high and low elevation classes samples late spring samples (******) were pooled over the period 2003 to 2006. To specifically assess lowland wetlands (low elevation class) conventional and organic farm sub-classes were compared to reference/non-grazing sites. Reference sites ($^{++}$) were a sub-sample of lowland sites classified as non-grazing and protected grazing. Where available, the Canadian Water Quality Guidelines (CWQG) for the protection of aquatic life is provided.

Site Classification	**Year**	**Sample Timing**	**Site N_{total}**	**Water Chemistry Parameters**															
				BOD	CL	F	SO_4	Br	NO_2	NO_3	PO_4	pH	Cond	Turb.	NH_3	N-total	o-PO_4 diss	P-Diss.	P-total
Units				mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	PH units	US/cm	NTU	mg/L	mg/L	mg/L	mg/L	mg/L
Detection Limit				5.0	0.5	0.02	3.0	0.05	0.002	0.005	0.05	0.01	2	0.05	0.005	0.02	0.001	0.004	0.004
CWQG (over long-term)				NA	120	0.12	NA	NA	0.06	13.0	NA	6.5 - 9.0	NA	NA	1.37	NA	NA		
Land-use Sub-class																			
Conventional Agriculture	**Pool***	**Late**	**42**	**8.1 (5.6)**	**21.0 (13.7)**	**0.53 (0.39)**	**197.0 (373)**	**0.07 (0.06)**	**0.048 (0.134)**	**0.009 (0.028)**	**0.06 (0.03)**	**8.36 (0.46)**	**753 (680)**	**6.57 (9.43)**	**0.092 (0.232)**	**1.76 (2.03)**	**0.030 (0.058)**	**0.052 (0.069)**	**0.126 (1.99)**
	2003	Late	11	5.4 (0.9)	21.6 (13.8)	0.52 (0.43)	168.7 (293.3)	0.07 (0.04)	0.105 (0.240)	< 0.005 (0)	0.07 (0.05)	8.27 (0.51)	741 (611)	3.72 (2.24)	0.120 (0.206)	1.55 (1.07)	0.048 (0.083)	0.071 (0.094)	0/130 (0.165)
	2004	Early	2	6.0 (1.4)	61.0 (50.9)	0.6 (0.31)	587.5 (753.1)	0.15 (0.13)	< 0.002 (0)	< 0.005 (0)	< 0.05 (0)	8.65 (0.21)	1473 (1453)	9.62 (1.67)	0.009 (0.005)	1.90 (1.41)	0.002 (0.001)	0.020 (0.012)	0.087 (0.069)
		Late	12	10.3 (9.7)	17.0 (14.4)	0.52 (0.35)	170.8 (305.3)	0.06 (0.05)	0.050 (0.087)	< 0.005 (0)	0.06 (0.02)	8.17 (0.44)	622 (540)	8.87 (14.42)	0.043 (0.069)	2.42 (3.50)	0.026 (0.045)	0.048 (0.058)	0.184 (0.328)
	2005	Early	3^	12.3 (4.6)	33.8 (19.5)	0.52 (0.60)	415.2 (606.2)	0.12 (0.10)	0.016 (0.014)	0.010 (0.012)	< 0.05 (0)	8.28 (0.16)	1381 (1293)	8.38 (8.63)	2.103 (3.213)	3.32 (3.59)	0.064 (0.082)	0.087 (0.094)	0.143 (0.090)
		Late	11	10.0 (0)	23.5 (15.1)	0.53 (0.39)	220.3 (446.4)	0.07 (0.08)	0.022 (0.046)	0.016 (0.035)	< 0.05 (0)	8.45 (0.33)	838 (797)	6.46 (6.60)	0.159 (0.401)	1.48 (0.93)	0.012 (0.026)	0.033 (0.034)	0.076 (0.049)
	2006	Early	3^	9.0 (3.3)	30.4 (14.6)	0.13 (0.21)	393.7 (522.6)	0.11 (0.04)	0.050 (0.103)	< 0.005 (0)	< 0.05 (0)	8.53 (0.22)	1169 (1092)	7.57 (6.74)	0.028 (0.028)	1.62 (1.13)	0.061 (0.094)	0.087 (0.102)	0.139 (0.108)
		Late	8	5.6 (1.2)	22.7 (11.1)	0.54 (0.48)	243.1 (503.9)	0.08 (0.07)	0.003 (0.001)	0.012 (0.021)	0.06 (0.01)	8.64 (0.46)	848 (870)	7.18 (10.10)	0.031 (0.029)	1.53 (0.94)	0.036 (0.069)	0.057 (0.082)	0.010 (0.096)
Organic Agriculture	**Pool***	**Late**	**23**	**7.35 (2.6)**	**30.7 (35.6)**	**0.45 (0.26)**	**650.0 (1118)**	**0.07 (0.05)**	**1.370 (3.38)**	**0.011 (0.025)**	**0.06 (0.04)**	**8.24 (0.62)**	**1302 (1407)**	**3.22 (2.74)**	**0.050 (0.095)**	**2.27 (3.17)**	**0.058 (0.138)**	**0.077 (0.154)**	**0.127 (0.195)**
	2003	Late	5	< 5.0 (0)	33.0 (40.4)	0.40 (0.22)	773.0 (1443.8)	0.08 (0.06)	2.655 (5.895)	0.029 (0.053)	< 0.05 (0)	8.18 (0.52)	1385 (1407)	4.24 (2.63)	0.043 (0.017)	3.57 (5.83)	0.010 (0.011)	0.021 (0.013)	0.058 (0.036)
	2004	Early	3	6.0 (1.7)	46.2 (57.1)	0.31 (0.27)	1207.7 (1558.2)	0.08 (0.06)	0.008 (0.010)	< 0.005 (0)	0.20 (0.27)	7.68 (0.18)	1899 (1038)	4.05 (2.67)	0.073 (0.038)	1.57 (1.07)	0.226 (0.384)	0.263 (0.422)	0.372 (0.456)
		Late	7	6.9 (2.5)	27.5 (28.2)	0.46 (0.24)	534.3 (680.1)	0.06 (0.03)	1.680 (3.117)	0.008 (0.008)	0.08 (0.07)	7.97 (0.53)	1185 (1011)	3.21 (2.37)	0.022 (0.025)	2.71 (2.43)	0.061 (0.112)	0.088 (0.133)	0.173 (0.223)
	2005	Early	1^	13.0 (4.2)	7.9 (2.3)	0.40 (0.07)	137.5 (89.9)	< 0.05 (0)	0.010 (0.011)	< 0.005 (0)	< 0.05 (0)	7.79 (0.23)	830 (66)	5.53 (2.01)	0.082 (0.039)	1.05 (0.21)	0.065 (0.062)	0.090 (0.069)	0.169 (0.099)
		Late	8	10.1 (0.8)	23.8 (35.9)	0.49 (0.28)	482.9 (1127.9)	0.07 (0.05)	0.820 (2.311)	< 0.005 (0)	< 0.05 (0)	8.18 (0.66	1098 (1426)	3.29 (3.53)	0.084 (0.159)	1.58 (2.06)	0.105 (0.209)	0.125 (0.231)	0.166 (0.255)
	2006	Early	2^	8.3(2.6	60.0	0.09	1758.0	1.05	0.006	< 0.005	< 0.05	8.17	2875	3.31	0.032	0.78	0.007	0.023	0.050

				)	(57.8)	(0.15)	(1718.4)	(1.97)	(0.006)	(0)	(0)	(0.15)	(2218)	(2.23)	(0.034)	(0.18)	(0.009)	(0.012)	(0.042)
		Late	3	5.0 (0)	52.9 (53.3)	0.38 (0.39)	1162.0 (1774.6)	0.09 (0.08)	< 0.002 (0)	< 0.005 (0)	< 0.05 (0)	8.94 (0.61)	1984 (2409)	1.38 (0.36)	0.038 (0.019)	0.92 (0.09)	0.002 (0.002)	0.018 (0.012)	0.031 (0.010)
Reference[++]	**Pool**	**Late**	**20**	**8 (2.6)**	**31.3 (40.4)**	**0.73 (0.32)**	**93.0 (65.6)**	**< 0.05 (0)**	**< 0.002 (0.099)**	**< 0.005 (0)**	**0.07 (0.08)**	**8.31 (0.45)**	**627 (205)**	**2.55 (1.733)**	**0.030 (0.03)**	**1.40 (0.14)**	**0.04 (0.138)**	**0.065 (0.153)**	**0.099 (0.190)**
Non-grazing	**Pool***	**Late**	**18**	**7.2 (2.5)**	**31.6 (42.7)**	**0.71 (0.35)**	**78.5 (62.1)**	**< 0.05 (0)**	**0.186 (0.540)**	**< 0.005 (0)**	**< 0.05 (0)**	**8.39 (0.43)**	**574 (239)**	**2.81 (1.73)**	**0.022 (0.030)**	**1.33 (0.70)**	**0.009 (0.017)**	**0.031 (0.031)**	**0.061 (0.048)**
	2003	Late	1	< 5.0	13.0	0.20	56.0	< 0.05	< 0.002	< 0.005	< 0.05	8.58	667	5.34	< 0.005	1.44	< 0.001	0.021	0.054
	2004	Late	6	6.5 (3.2)	42.5 (63.9)	0.82 (0.30)	77.0 (44.3)	< 0.05 (0)	0.554 (0.864)	< 0.005 (0)	< 0.05 (0)	8.47 (0.27)	529 (188)	2.80 (2.01)	0.013 (0.009)	1.06 (0.39)	0.019 (0.025)	0.043 (0.039)	0.063 (0.039)
	2005	Early	1^	11.5 (2.1)	26.0 (8.5)	0.42 (0.01)	115.0 (22.6)	<0.05 (0)	0.009 (0.009)	< 0.005 (0)	< 0.05 (0)	8.25 (0.09)	744 (2)	0.95 (0.51)	0.009 (0.003)	0.98 (0.21)	0.004 (0.002)	0.028 (0.018)	0.039 (0.025)
		Late	5	8.0 (2.7)	37.4 (43.7)	0.67 (0.36)	69.7 (49.0)	< 0.05 (0)	< 0.002 (0)	< 0.005 (0)	< 0.05 (0)	8.19 (0.29)	591 (146)	2.57 (1.85)	0.038 (0.051	1.45 (1.10)	< 0.001 (0)	0.021 (0.016)	0.055 (0.055)
	2006	Early	1^	10.5 (2.1)	20.3 (3.6)	0.32 (0.44)	142.5 (7.8)	< 0.05 (0)	0.010 (0.008)	< 0.005 (0)	< 0.05 (0)	8.25 (0.03)	869 (30)	2.19 (0.69)	0.007 (0.002)	0.88 (0.11)	0.003 (0.001)	0..034 (0.006)	0.055 (0.006)
		Late	6	7.5 (1.8)	19.1 (12.8)	0.73 (0.39)	90.9 (94.2)	< 0.05 (0)	< 0.002 (0)	< 0.005 (0)	8.44 (0.66)	5.90 (371)	2.63 (1.40)	0.02 (0.022)	1.49 (0.66)	0.008 (0.013)	0.008 (0.013)	0.031 (0.035)	0.064 (0.061)
Protected Grazing	**Pool***	**Late**	**6**	**5.2 (0.4)**	**40.3 (24.9)**	**0.38 (0.32)**	**1804.0 (1371.0)**	**0.41 (0.30)**	**0.037 (0.074)**	**< 0.005 (0)**	**< 0.05 (0.01)**	**9.01 (0.66)**	**3514 (2270)**	**4.65 (4.22)**	**0.028 (0.020)**	**2.85 (1.65)**	**0.022 (0.033)**	**0.056 (0.047)**	**0.083 (0.044)**
	2004	Late	3	5.3 (0.6)	33.4 (16.1)	0.55 (0.22)	1814.3 (1534.3)	0.35 (0.27)	0.072 (0.100)	< 0.005 (0)	0.06 (0.01)	9.00 (0.77)	3148 (2145)	4.81 (5.88)	0.033 (0.026)	2.86 (1.75)	0.026 (0.043)	0.077 (0.062)	0.095 (0.057)
	2005	Late	3	< 5.0 (0)	47.2 (33.8)	0.21 (0.35)	1793.7 (1533.4)	0.47 (0.37)	< 0.002 (0)	< 0.005 (0)	< 0.05 (0)	9.02 (0.69)	3880 (2807)	4.50 (3.14)	0.0222 (0.015)	2.84 (1.93)	0.017 (0.028)	0.034 (0.017)	0.071 (0.032)
Unprotected Grazing	**Pool***	**Late**	**23**	**15.5 (18.2)**	**112.3 (267.9)**	**0.37 (0.29)**	**819.8 (1733.7)**	**0.28 (0.89)**	**0.239 (1.039)**	**< 0.005 (0.001)**	**0.12 (0.23)**	**8.06 (0.49)**	**1953 (3277)**	**13.26 (45.28)**	**0.170 (0.683)**	**1.83 (2.64)**	**0.140 (0.408)**	**0.202 (0.438)**	**0.381 (0.888)**
	2004	Late	5	25.4 (38.5)	109.4 (203.0)	0.33 (0.31)	1824.1 (2366.5)	0.90 (1.90)	1.039 (2.216)	< 0.005 (0)	0.12 (0.16)	8.12 (0.82)	4212 (5802)	45.84 (94.58)	0.678 (1.466)	4.24 (5.14)	0.126 (0.276)	0.335 (0.377)	0.999 (1.676)
	2005	Early	1^	11.0 (5.7)	24.6 (10.5)	0.44 (0.13)	123.0 (14.1)	< 0.05 (0)	0.007 (0.007)	< 0.005 (0)	8.52 (0.73)	8.52 (0.73)	714 (134)	2.14 (1.12)	1.08 (0.311)	1.08 (0.311)	0.004 (0.004)	0.028 (0.016)	0.053 (0.028)
		Late	10	8.1 (2.5)	126.1 (329.7)	0.34 (0.27)	527.1 (1540.6)	0.10 (0.15)	0.013 (0.033)	0.006 (0.001)	0.16 (0.33)	7.94 (0.35)	1265 (2102)	2.85 (2.90)	0.028 (0.016)	0.99 (0.48)	0.215 (0.588)	0.237 (0.599)	0.261 (0.609)
	2006	Early	1^	9.5 (0.7)	24.4 (2.3)	0.19 (0.25)	166.0 (2.8)	0.08 (0.04)	0.009 (0.008)	< 0.005 (0)	< 0.05 (0)	8.28 (0.08)	868 (100)	1.14 (0.47)	0.010 (0.001)	0.66 (0.01)	< 0.001 (0)	0.012 (0.001)	0.127 (0.146)
		Late	8	10.0 (5.4)	96.8 (248.5)	0.44 (0.32)	558.1 (1508.0)	0.11 (0.14)	0.020 (0.035)	< 0.005 (0)	0.07 (0.06)	8.18 (0.40)	1401 (2039)	4.84 (7.25)	0.031 (0.050)	1.38 (1.07)	0.056 (0.138)	0.076 (0.163)	0.144 (0.201)
Residential	**Pool***	**Late**	**5**	**9.2**	**29.1 (6.24)**	**0.24 (0.21)**	**72.2 (23.2)**	**0.06 (0.02)**	**0.037 (0.062)**	**0.015 (0.023)**	**< 0.05 (0.01)**	**8.22 (0.31)**	**594 (211)**	**11.46 (8.74)**	**0.069 (0.100)**	**1.82 (0.85)**	**0.136 (0.216**	**0.165 (2.37)**	**0.317 (0.307)**
	2004	Late	1	8.0	29.0	0.43	67.0	< 0.05	0.022	< 0.005	< 0.05	8.48	342	4.05	0.006	0.95	< 0.001	0.005	0.028
	2005	Late	3	10.3 (0.6)	25.8 (3.6)	0.25 (0.21)	70.3 (31.6)	0.06 (0.02)	0.053 (0.080)	0.022 (0.030)	< 0 .05 (0)	8.23 (0.34)	589 (149)	10.74 (8.89)	0.109 (0.119)	1.85 (0.88)	0.166 (0.286)	0.194 (0.308)	0.319 (0.327)
	2006	Late	1	7.0	39	0.01	83.0	< 0.05	< 0.002	< 0.005	0.08	7.95	858	21.00	0.014	2.60	0.182	0.239	0.600
Golf Course	**Pool***	**Late**	**4**	**8.0 (2.9)**	**234.0 (155.0)**	**0.33 (0.08)**	**289.0 (219)**	**0.15 (0.13)**	**4.814 (3.51)**	**0.066 (0.108)**	**< 0.05 (0)**	**8.12 (0.08)**	**1699 (825)**	**10.84 (6.13)**	**0.141 (0.073)**	**6.3 (2.89)**	**0.005 (0.008)**	**0.053 (0.069)**	**0.189 (0.119)**
	2004	Late	2	8.0 (4.2)	121.8 (147.4)	0.32 (0.10)	102.5 (68.6)	0.07 (0.02)	2.038 (2.181)	0.016 (0.016)	< 0.05 (0)	8.06 (0.06)	1068 (668)	14.35 (3.75)	0.100 (0.076)	3.85 (0.35)	< 0.001 (0)	0.001 (0.006)	0.158 (0.098)
	2005	Late	1	10.0	347.0	0.26	496.0	0.34	8.400	< 0.005	< 0.05	8.18	2290	2.37	0.140	9.40	0.014	0.035	0.098
	2006	Late	1	6.0	346.0	0.41	454.0	0.11	6.780	0.227	< 0.05	8,17	2370	12.30	0.224	8.10	0	0.166	0.340
Elevation Sub-class																			
Low Elevation:	**Pool****	**Late**	**94**	**7.9 (4.2)**	**35.1 (57.4)**	**0.53 (0.35)**	**283.0 (637.3)**	**0.07 (0.05)**	**0.586 (2.066)**	**0.012 (0.029)**	**0.06 (0.05)**	**8.30 (0.48)**	**893 (896)**	**5.34 (7.27)**	**0.068 (0.166)**	**2.334 (2.345)**	**0.044 (0.113)**	**0.067 (0.126)**	**0.133 (0.201)**

Valley Lowlands	2003	Late	17	3.1 (1.5)	24.5 (23.7)	0.47 (0.37)	339.8 (811.6)	0.04 (0.06)	0.848 (3.189)	0.008 (0.030)	0.03 (0.05)	8.30 (0.5)	926 (966)	3.97 (2.25)	0.090 (0.168)	2.141 (3.184)	0.034 (0.069)	0.053 (0.079)	0.104 (0.136)
	2004	Early	5	5.0 (2.5)	52.1 (48.4)	0.41 (0.28)	959.6 (1212.9)	0.09 (0.10)	0.005 (0.008)	0 (0)	0.11 (0.23)	8.07 (0.55)	1728 (1568)	6.28 (3.68)	0.047 (0.044)	1.702 (1.049)	0.136 (0.298)	0.164 (0.328)	0.258 (0.360)
		Late	29	7.8 (7.2)	32.6 (49.6)	0.56 (0.32)	233.2 (409.7)	0.03 (0.04)	0.635 (1.725)	0.002 (0.007)	0.040 (0.087)	8.15 (0.45)	770 (655)	6.21 (9.88)	0.034 (0.052)	2.272 (3.581)	0.049 (0.127)	0.073 (0.143)	0.171 (0.275)
	2005	Early	6^	12.1 (3.9)	26.7 (16.8)	0.47 (0.41)	270.2 (436.8)	0.06 (0.09)	0.012 (0.012)	0.003 (0.010)	0.006 (0)	8.23 (0.34)	1071 (931)	5.63 (6.79)	1.073 (2.418)	2.179 (2.701)	0.044 (0.065)	0.069 (0.074)	0.115 (0.086)
		Late	30	3.9 (3.0)	36.6 (64.3)	0.51 (0.33)	252.1 (635.5)	0.04 (0.08)	0.512 (1.907)	0.006 (0.024)	0.007 (0.008)	8.26 (0.44)	882 (906)	4.91 (5.59)	0.106 (0.255)	1.784 (1.908)	0.052 (0.141)	0.074 (0.153	0.122 (0.175)
	2006	Early	7^	8.9 (3.0)	36.5 (33.2)	0.16 (0.22)	715.1 (1125.0)	0.35 (1.05)	0.025 (0.068)	0 (0)	0.006 (0)	8.35 (0.23)	1570 (1531)	4.66 (5.10)	0.023 (0.026)	1.135 (0.829	0.028 (0.065)	0.050 (0.072)	0.100 (0.092)
		Late	18	5.1 (2.6)	46.6 (78.3)	0.56 (0.43)	361.1 (788.1)	0.05 (0.07)	0.378 (1.598)	0.017 (0.055)	0.015 (0.026)	8.63 (0.49)	1073 (1158)	5.94 (8.14)	0.039 (0.052)	1.811 (1.756)	0.029 (0.061)	0.058 (0.079)	0.118 (0.155)
High Elevation: Valley Uplands	**Pool****	**Late**	**27**	**10.7 (17.1)**	**102.5 (247.8)**	**0.34 (0.28)**	**1082.7 (1740,9)**	**0.32 (0.83)**	**0.262 (0.982)**	**< 0.005 (0.001)**	**0.096 (0.203)**	**8.31 (0.66)**	**2363 (3230)**	**12.28 (41.59)**	**0.147 (0.631)**	**2.049 (2.574)**	**0.101 (0.365)**	**0.158 (0.393)**	**0.310 (0.818)**
	2004	Late	7	16.6 (34.2)	89.5 (169.4)	0.35 (0.26)	2046.1 (2010.0)	0.76 (1.58)	0.966 (1.843)	0 (0)	0.021 (0.027)	8.66 (0.79)	4228 (4770)	34.58 (79.64)	0.494 (1.237)	3.851 (4.405)	0.019 (0.031)	0.181 (0.271)	0.639 (1.452)
	2005	Late	11	2.9 (1.0)	124.0 (312.4)	0.28) (0.28)	948.4 (1690.4)	0.19 (0.29)	0.011 (0.032	0.002 (0.003)	0.105 (0.330)	8.25 (0.671)	2075 (2612)	3.61 (3.02)	0.022 (0.011	1.463 (1.309)	0.194 (0.562)	0.213 (0.573)	0.241 (0.582)
	2006	Late	9	9.6 (5.2)	86.2 (234.6)	0.41 (0.30)	497.5 (1422.3)	0.08 (0.15)	0.017 (0.034	0 (0)	0.029 (0.068)	8.10 (0.45)	1264 (1950)	5.44 (6.69)	0.029 (0.047)	1.364 (0.997)	0.051 (0.130)	0.073 (0.153)	0.139 (0.189)

^ Sites sampled twice in early spring (e.g. Elevation sub-class: 2005, early sample, six lowland valley sites were sampled twice, totaling 12 water samples)

Table 3.2. Wetland monitoring (2003 to 2006) targeted eight amphibian and one turtle species in the south Okanagan Valley, B.C.

Common Name	**Scientific Name**	**Species Acronym**	**Order**	**Family**	**Authority**	**Provincial COSEWIC Designation**
Blotched tiger salamander	*Ambystoma mavortium melanostictum*	A-AMMV	Caudata	Ambystomatidae	Baird, 1850	Endangered, 2001
Long-toed salamander	*Ambystoma macrodactylum*	A-AMMA	Caudata	Ambystomatidae	Baird, 1849	Not listed
Great Basin spadefoot	*Spea intermontana*	A-SPIN	Anura	Scaphiopodidae	Cope, 1883	Threatened, 2001
Western toad	*Bufo boreas*	A-BUBO	Anura	Bufonidae	Baird and Girard, 1852	Special Concern, 2002
Columbia spotted frog	*Rana luteiventris*	A-RALU	Anura	Ranidae	Thompson, 1913	Not listed
Pacific chorus frog	*Pseudacris regilla*	A-PSRE	Anura	Hylidae	Baird and Girard, 1852	Not listed
Northern leopard frog	*Lithobates pipiens*	A-RAPI	Anura	Ranidae	Schreber, 1782	Endangered, 2000; Locally extirpated
American bullfrog	*Lithobates catesbeiana*	A-LICA	Anura	Ranidae	Shaw, 1802	Non-native species
Western painted turtle (Inter-mountain-Rocky Mountain)	*Chrysemys picta bellii*	R-CHPI	Testudines	Emydidaem	Schneider, 1783	Locally: Special Concern, 2006

3.2.5 Wetland surveys

Surveys were conducted annually from 2003 to 2006; due to logistics and variation in species phonology the earliest survey conducted in any given year was 18 April and latest survey date was 17 July. The same observer (author S. Ashpole) conducted the monitoring in each year, and on most occasions a second observer assisted with monitoring. To determine species richness, distribution, and relative abundance of amphibian species, auditory, time-constrained visual encounter search, and nighttime trapping surveys were conducted. These three survey types were recommended by the B.C. Ministry of Environment during the amphibian-breeding season to detect reproductive adults, egg masses, and tadpoles or larvae (B.C. Ministry of Environment, Lands, and Parks, 1998). Western painted turtle (*Chrysemys picta bellii)* observations were opportunistic visual sightings. Amphibian and turtle observations were assessed for species identification (Corkran and Thoms, 1996), reproductive stage of development (for amphibians; Gosner, 1960), and any indication of injury or deformity. With the exception of the American bullfrog, all species were released at the site of capture. American bullfrogs were humanely killed. The date, time of observation, duration of survey, Global Positioning System (GPS) coordinates, and weather conditions were recorded.

All procedures conducted in this research followed the Canadian Council on Animal Care (Olfert et al., 1993) using approved protocols from Environment Canada (Delta, B.C.) and Simon Fraser University (AUP # 730B04) Animal Care Committees; research permits were obtained from the BC Ministry of Environment (# PE06-21835). To ensure no cross contamination of disease (e.g. *Batrachochytrium dendrobatidis*) or transport of non-native species among sites all field equipment was disinfected with 10% bleach solution daily. To further reduce risk, sites with known American bullfrog populations had dedicated site-specific equipment.

3.2.5.1 Visual encounter survey

Daytime visual encounter surveys were used to determine relative abundance of species and life stage (e.g. egg, larval, metamorphic, hatchling, adult) using a time-constrained effort. Daytime surveys were conducted a minimum of three occasions at each site surveyed per year. All species observed were counted and the life stage recorded. Each zone of vegetation (e.g. emergent, sub-emergent, open water) was surveyed for species along a curvilinear transect encircling the pond or wetland system (B.C. Ministry of Environment, Lands, and Parks, 1998). The observers timed their effort while visually searching and dip-netting each site. In the case of large open water wetlands or where private access was limited (N = 18) searching was restricted to accessible portions of the site. It is acknowledged that the inference of species detection is limited to the area surveyed only. The difference between surveying a sites in entirety versus a restricted area is not accommodated in analysis and may pose a sampling bias.

3.2.5.2 Auditory survey

Listening for the breeding calls of male frogs was used to further evaluate presence or not-detected occurrences of calling amphibian species. Auditory surveys were conducted after sunset on three to ten occasions annually at each discrete wetland site in that year. Additionally, the Okanagan River Channel dike trail (approximately 8 km transect from the head of Osoyoos Lake to North of Oliver) was surveyed by foot on three occasions in 2006; all calling individuals and the direction of the calling were recorded. The actual location and aquatic habitat type (e.g. ephemeral wetlands, ponds, and old oxbows bisected by the channel) could not be determined in most cases. The time for an individual auditory survey at both discrete pond sites and non-discrete sites along the river channel were conducted for at least five (uninterrupted) minutes. Auditory observations were recorded using a calling index, where 0 = no calls heard, 1 = calling individuals can be counted, 2 = calls of individuals are distinguishable (some calls overlap), 3 = full chorus, individuals cannot be distinguished (B.C. Ministry of Environment, Lands, and Parks, 1998).

3.2.5.3 Night-time trapping survey

Trapping surveys were used to target salamanders as well as other amphibian species. To reduce capture stress on reproducing amphibian species, night-time trapping was initiated in late May at sites post peak amphibian reproductive activity. Minnow traps with foam floats attached were deployed and baited with canned fish, and placed floating in the emergent vegetation zone of the site. Three to six traps were set approximately 1 to 2 hours before sunset and were removed the following morning. Trapping effort was quantified by trap night, where trap night = number of traps × number nights trapped (B.C. Ministry of Environment, Lands, and Parks, 1998). Trapping surveys were conducted three to five times per site and annual trapping effort ranged from 9 to 30 trap nights per site, with the exception of two sites that had American bullfrogs, where the annual trapping effort ranged from 170 to 2320 trap nights a site per year.

3.2.6 Statistical analysis

Parametric statistical analysis was conducted where the data generally met requirements (normal distribution, homoscedasticity). Parametric tests are acceptably robust even if there are moderate deviations from their assumptions e.g. non-normality when minimal skewing (Johnson 1995). Where conditions were not met a non-parametric statistical approach is indicated. Observation data were pooled across years because (a) the frequency of detection of most species was too low to perform comparisons among years and (b) all sites were not surveyed annually (i.e. limited property access, absence of water, and increased site detection with time). All statistical analyses were performed using Statistica 6.1 (Statsoft, 2003).

3.2.6.1 Habitat parameters

Because some land-use sub classes were few in number, statistical analysis comparing pond perimeter, pond depth, distance from high water mark to nearest crop and nearest road was limited to conventional agriculture, organic agriculture, and unprotected grazing sites with other land-use sub classes were omitted from the analysis (Golf course N = 1 site, Residential N =1, Protected Grazing N = 3). Analysis of variance (ANOVA) and F-test (normal distributions and homogeneity of variances) was used to assess each habitat parameter among conventional and organic agricultural sites, and unprotected grazing sites. Additionally descriptive comparisons were made with water permanency and elevation, irrespective of land-use sub class. Analysis correlating habitat parameters and the detection of species was not made due the limitation of a single year of data on habitat features and the extent of observed habitat modifications over the course of the study.

3.2.6.2 Water chemistry

Water chemistry was compared among wetland classes, which were classified *a priori* into seven categories according to land-use practices (sub class: conventional farm, organic farm, reference, protected grazing, unprotected grazing, golf course, residential). All wetland sites were classified as either lowland or upland (elevation class) and by sampling period (as either early or late water sampling). To specifically assess lowland wetlands (low elevation class) in agricultural land-use (main class), the conventional farm and organic farm sub-classes were compared to reference sites. Reference sites were a sub-sample of lowland sites classified as non-grazing and protected grazing.

Annual sample sizes were low and therefore water chemistry results were pooled across years by site within sampling period. Parameter values below the analytical minimal detection limits (MDL) were substituted with the proportion of observations above the MDL × MDL (e.g. for fluorine which has 85.2% observations above MDL, substitution values were 0.852 × 0.02 mg/L F) (McCarthy et. al., 1997). Phosphorous (o-PO_4 dissolved) and total dissolved phosphorus were highly redundant ($R^2 = 0.90$, $F_{(1, 150)} = 1317.7$, $P < 0.001$), consequently the latter was excluded from analysis. Phosphorous was retained due to its biological availability. A multivariate discriminant function analysis (DFA) was used to determine the degree that wetland land-use sub-classes differ and which water chemistry parameters were driving the differences. DFA is used when groups (e.g. land-use classifications) are known *a priori* and can be used with small sample sizes. Factor analysis (FA) was used to examine the relationships among water quality variables, using a maximum likelihood extraction and varimax normalized rotation for the FA. Factors were included if the eigenvalues were greater than 1.0 prior to rotation (i.e. the factor contributed more variance than a single variable).

3.2.6.3 Species richness

Species richness data were quantified using the data collected only at discrete wetland sites where multi-survey methods allowed for the detection of all possible target species, and excluded sites with auditory only records (e.g. river channel transect) and incidental observations. Analysis of variance (ANOVA) was used to assess species richness among categories with 1) main and sub-class land-use, 2) water permanency, 3) elevation, and 4) presence of non-native invasive fish. ANOVAs were used because the analysis does not require the raw data to be normally distributed (Johnson 1995). Due to small sample sizes the analysis examining the effect of the presence of fish on amphibian species richness was restricted to vacant sites (N = 77 sites) and sites where non-native invasive fish were detected (N = 29 sites), and excluded sites with native fish (N = 2 sites) and unknown sites (N = 6 sites). Similarly, due to small sample sizes, the analysis by land-use sub-class excluded residential sites (N = 2 records), golf courses (N = 2 records), artificial pools (N = 5 records), and protected grazing sites (N = 5 records). In addition, a second analysis examining the presence of fish on species richness was conducted to determine if frogs calling at sites were reproductively successful i.e. discrete wetland sites with known early life stages recorded. To determine species that are likely to coexist a cluster analysis was used (unweighted pair-group, using 1-pearson r).

3.2.6.4 Auditory estimates of population density: Pacific chorus frog and Great Basin spadefoot

Auditory calling data from wetland sites and non-discrete sites (e.g. river channel transect) was used to assess the auditory calling index (0 to 3) for relative population density of the Great Basin spadefoot (*Spea intermontana*) (generally short calling period with explosive breeding patterns and asynchronous) and Pacific chorus frog (*Pseudacris regillia*) (generally long calling asynchronous and breeding period). Auditory indices are commonly used as an indictor of population density (Weir et al., 2005). Both our observer bias was inherently reduced by minimizing the number of observers and our species detectably was high due the low number of calling species to detect (de Solla et al., 2006). Data excluded from the analysis included incidental auditory observations and calling records from the western toad (non-calling species; Black and Brunson, 1971), Columbia spotted frog (inaudible species), and the America bullfrog (low sample size, N = 28 records). A generalized linear model (GLZ: for background, *see* McCullagh and Nelder 1989) was used to evaluate calling code intensity and the possible effect of land-use class (including the river channel class), sub-class, elevation class, water permanency class, and the presence of fish. Because the data were counts, a Poisson distribution was used with a log link function. Type III likelihood ratio tests were used to compare the deviance of the full model to the null model. Pearson χ^2 was used to detect over-dispersion, when the observed variance is greater than the predicted variance, or under dispersion, when the observed variance is less than the predicted variance. Dispersion was corrected by including a dispersion factor ϕ (Ef), and the covariance matrix was multiplied by ϕ, and the log likelihoods used in the likelihood ratio tests were divided by factor ϕ (McCullagh and Nelder 1989). Non-overlapping confidence intervals were used as a GLZ post hoc test to determine statistical significance between treatment classes and set at either 95% or 84% (approximate α of 0.05; Payton et al., 2003).

3.2.6.5 Relative density of early life stages

Monitoring data used to assess species relative density of early life stage as an indicator of reproductive success among sites included only discrete wetland sites where survey methods allowed for the detection of all possible early life stages (e.g. egg mass, tadpole, larvae, metamorph, or hatchling). A relative density of 0 to 4 was assigned to each species at a site and corresponded to the total number of early life stages observed (0 = none detected; 1 = 1 to 9 individuals; 2 = 10 to 99 individuals; 3 = 100 to 999 individuals; 4 = 1000 to a 3247 maximum of individuals observed). As with auditory estimates *(section 3.2.6.4*) a GLZ test and post hoc assessment was used to evaluate the effect of class, subclass, elevation, water permanency, and presence of fish on the relative density of each amphibian species.

3.3 RESULTS

3.3.1 Habitat parameters

Habitat characteristics of wetland habitats were different among and within subclasses, with few clear trends and large standard deviations among most results (Appendix 3.2A). The wetland perimeter significantly varied among land-use classifications with unprotected grazing sites, on average, 1.5 times larger than conventional and organic agricultural sites ($F_{(2, 31)} = 4.91$, $p = 0.01$). Similarly, the wetland pond depth at the center point significantly varied with land use, and conventional and organic agricultural sites were up to 3 times deeper compared to unprotected grazing sites ($F_{(2, 31)} = 10.62$, $p < 0.001$). Regardless of elevation, pond depths of temporary ponds were shallower compared to permanent ponds. There were no significant differences in the distance from the wetland to the nearest crop between conventional and organic agricultural sites ($F_{(1, 25)} = 2.29$, $p = 0.14$). The minimum distance from a wetland to a road varied significantly among conventional and organic agricultural sites, and this distance was longest at unprotected grazing sites ($F_{(2, 31)} = 8.23$, $p = 0.001$). Residential and conventional agricultural sites were located nearest to roads, whereas high elevation protected grazing sites were the furthest from roads.

The frequency of withdrawal or discharge, infilling, garbage, introduced invasive species, and agricultural input (e.g. pesticides, herbicides) was similar across all sites ranging from 14.8 to 19.4% (Appendix 3.2B). Anthropogenic nutrient input occurred in 59.2% of sites (N = 64 sites), and was limited to conventional (N = 16 sites), organic (N = 14 sites), unprotected grazing sites (N = 31 sites), and golf courses (N = 3 sites). The most common stressor that occurred across all sub-classes was the presence of introduced invasive species in 19% of sites (N = 21 sites) and the wetland was artificially constructed in 37% of sites (N = 40 sites). Of the seven anthropogenic stressors assessed, 88% of sites (N = 95 sites) had at least one and only 1.9% of sites (N = 2 sites) were subject to all seven (Appendix 3.2C). The highest frequency of stressors occurred at conventional and organic agricultural sites with the least impacted being protected grazing sites.

3.3.2 Water chemistry

Water quality parameters that exceeded the Canadian Water Quality Guidelines (CWQG) for the protection of aquatic life (Canadian Council of Ministers of the Environment. 2007) included levels of chloride, fluoride, nitrite, pH, and ammonium. The nitrite levels were highest among golf courses (8.4 mg mg NO_2^-/L.) and organic sites (2.7 mg NO_2^-/L.). Chloride concentration in grazed and golf course sites exceeded CWQG long-term acceptable levels (CWQG Cl 120 mg/L; Table 2). Chloride levels were more than double the guideline in samples from high elevation unprotected grazing sites (max. 1060 mg/L) and three samples from golf course sites (max. 346

mg/L). With the exception of eleven samples below detection limits, fluoride exceeded acceptable aquatic levels in all remaining water samples and ranged from 0.13 to 1.77 mg/L (CWQG F 0.12 mg/L; Table 3.1). Nitrite exceeded CWQG (CWQG NO2 0.06 mg/L; Table 3.1) in 23 samples and reached values as high as 13.2 mg/L among agricultural, golf course, and unprotected grazing sites. Ammonium exceeded CWQG (CWQG NH3 1.37 mg/L; Table 3.1) in two samples and reached 3.3 mg/L at one unprotected grazing sites in 2004. Lastly, pH exceeded acceptable water quality levels (CWQG pH 6 to 9; Table 3.1) on 18 occasions and reached pH values as high as 9.69 among agricultural and unprotected, protected and non-grazing sites.

3.3.2.1 Early vs. late season water sampling:

To compare water chemistry between early and late sampling events across all years DFA was used, including only sites that were sampled at both time periods (N = 8 sites). Water chemistry differed in six of 15 parameters between sample periods and could be partially differentiated by BOD, SO_4, conductivity, turbidity, NH_3, and N-total. Because of the differences between the early and late water sampling (DFA: Wilk's $\lambda = 0.45$, $F_{(15, 42)} = 3.38$, $p < 0.001$) all the subsequent analysis included only late water sampling events, which provide a larger more complete data set among sites (Table 3.1, *see* 137 late samples vs. 18 early samples).

3.3.2.2 Lowland vs. upland elevation water sampling:

Late water chemistry sampling differed in three of 15 parameters (Br, pH, P-total) between the low (N = 94 sites) and high elevation sites (N = 27 sites) (DFA: Wilk's $\lambda = 0.70$, $F_{(15, 105)} = 3.01$, $p < 0.001$). Elevation was omitted from subsequent analysis, because very few differences were found between elevation classes and only two of seven site classes (unprotected grazing and non-grazing) were found both in the lowland and upper valley.

3.3.2.3 Land-use and sub-class water sampling:

Water chemistry differed in three of 15 parameters (NO_2, pH, N-total) among the seven land-use classes (DFA: Wilk's $\lambda = 0.15$, $F_{(90, 568)} = 2.55$, $p < 0.0001$). In general, the protected grazing site was most dissimilar among all of the land-use classes. The pairs of categories that are most dissimilar are Golf Courses and Protected Grazing, followed by Protected Grazing and Residential, and Protected Grazing and Unprotected Grazing. The most similar are Conventional and Non-grazing (Table 3.3).

Table 3.3. Main land-use class comparisons of water chemistry samples using DFA showing significant ($p < 0.05$) Mahalanobis distances, south Okanagan Valley, B.C.

	Number samples in analysis	Organic Farm	Unprotected Grazing	Conventional Farm	Golf Course	Protected Grazing	Residential	Non-Grazing
Organic Farm	23							
Unprotected Grazing	23	**3.52***						
Conventional Farm	42	**2.77***	**3.49***					
Golf Course	4	**10.62***	**14.51***	**12.84***				
Protected Grazing	6	20.07	**20.75***	**16.88***	**29.89***			
Residential	5	4.78	5.96	3.49	**13.90***	**23.04***		
Non-grazing	18	2.14	3.77	1.53	**14.47***	**18.74***	6.38	

Bolded (*) values identify significance.

From the sub-classes DFA analysis, Root 1 was associated with higher pH, which was highest at the protected grazing sites relative to all the other sites. Root 2 was negatively associated with nitrate, nitrite and total nitrogen, which were highest at the golf course sites compared to the other sites (Appendix 3.3: Water chemistry (3A) DFA Root Structure, (3B) Main land-use class DFA factor structure; (3C) DFA root structure: Wilk's $\lambda = 0.22$, $F_{(90, 743)} = 2.57$, $p < 0.0001$).

<u>Conventional farm vs. organic farm, vs. reference land-use sub-classes:</u>

Water chemistry differed in six of 15 parameters (Cl, F, Br, NO_2, N-total, P-total) among conventional farm, organic farm, and reference site classes (DFA: Wilk's $\lambda = 0.11$, $F_{(30, 136)} = 3.59$, $p < 0.0001$). Based upon the Squared Mahalanobis Distances (SMD), the organic and reference sites were the pair of sites most dissimilar (SMD = 6.50, $F_{(15, 68)} = 3.84$, $p < 0.001$), whereas the reference sites and conventional sites were the least dissimilar, though still significantly different from each other (SMD = 3.98, $F_{(15, 68)} = 2.98$, $p < 0.001$) (Table 3.4). From the agricultural versus reference land-use classes DFA analysis, Root 1 was associated with higher SO_4 and NO_2 at the organic sites relative to conventional and reference sites and Root 2 was negatively associated with turbidity (Appendix 3.4: A DFA Root Structure, B subclass DFA factor structure).

Table 3.4. Comparisons of water chemistry among wetlands from agricultural and reference sub-class land-uses by DFA and significant ($p < 0.05$) Mahalanobis distances, south Okanagan Valley, B.C.

Sub class	Number samples in analysis	Reference	Conventional Farm	Organic Farm
Reference	20			
Conventional Farm	42	**3.98***		
Organic Farm	23	**6.50***	**4.93***	

Bolded (*) values identify significance.

Using Factor Analysis, four factors were extracted from the water chemistry, which accounted for 26.8, 19.0, 15.2, and 11.4% of the variance after varimax rotation (Appendix 3.5); 72.4% of the variance was explained in total. The first factor had large positive loadings for SO_4 and conductivity. Factor 2 had large positive loadings with PO_4, o-PO_4-diss, diss-P phosphorus, and a weak positive loading with P-total. Factor 3 contributed the most to the variance observed, and had large positive loadings for BOD, Br, turbidity, NH_3, and P-total, and a weak positive loading for N-total, while the fourth factor had large positive loadings for NO_2, NO_3, and N-total.

3.3.3 Species monitoring

With the exception of the locally extirpated Northern leopard frog, all target species were observed in the study area (Fig 3.3; Appendix 3.6A), with the largest number of observations recorded for the Pacific chorus frog (961 occurrences, Appendix 3.6B) and the Great Basin spadefoot (498 occurrences, Appendix 3.6C). Actively searching sites yielded more observations compared to all other survey methods. Salamander species were most likely observed when trapping, while species with an audible breeding call (Great Basin spadefoot and Pacific chorus frog) were frequently detected by auditory survey (Table 3.5). A cluster analysis grouped species, for example Columbia spotted frogs and Western toads, which were likely associated together (Fig. 3.4).

Table 3.5. Summary of species counts (records used for statistical analysis versus total occurrence) by survey method and incidental records, south Okanagan Valley, B.C., 2003 to 2006.

Species	Wetland survey method			Incidental observations		Total number observations
	Auditory	Visual encounter	Trapping	Auditory	Visual encounter	
Unknown *Ambystoma* salamander sp.	NC-sp	1	0	0	0	1
Western toad	6*	22	0	2	1	31
Long-toed salamander	NC-sp	27	35	0	1	63
Blotched tiger salamander	NC-sp	32	42	0	9	83
Columbia spotted frog	5*	74	6	0	3	88
Western painted turtle	NC-sp	53	4	0	85	142
American bullfrog	28	67	95	7	14	211
Great Basin spadefoot	141	107	78	10	14	350
Pacific chorus frog	376	192	209	36	22	835
Grand total	556	575	469	55	149	1804

(*Western toads rarely make an audible call, similarly spotted frogs call under the water and are rarely audible; NC-sp: non-calling species)

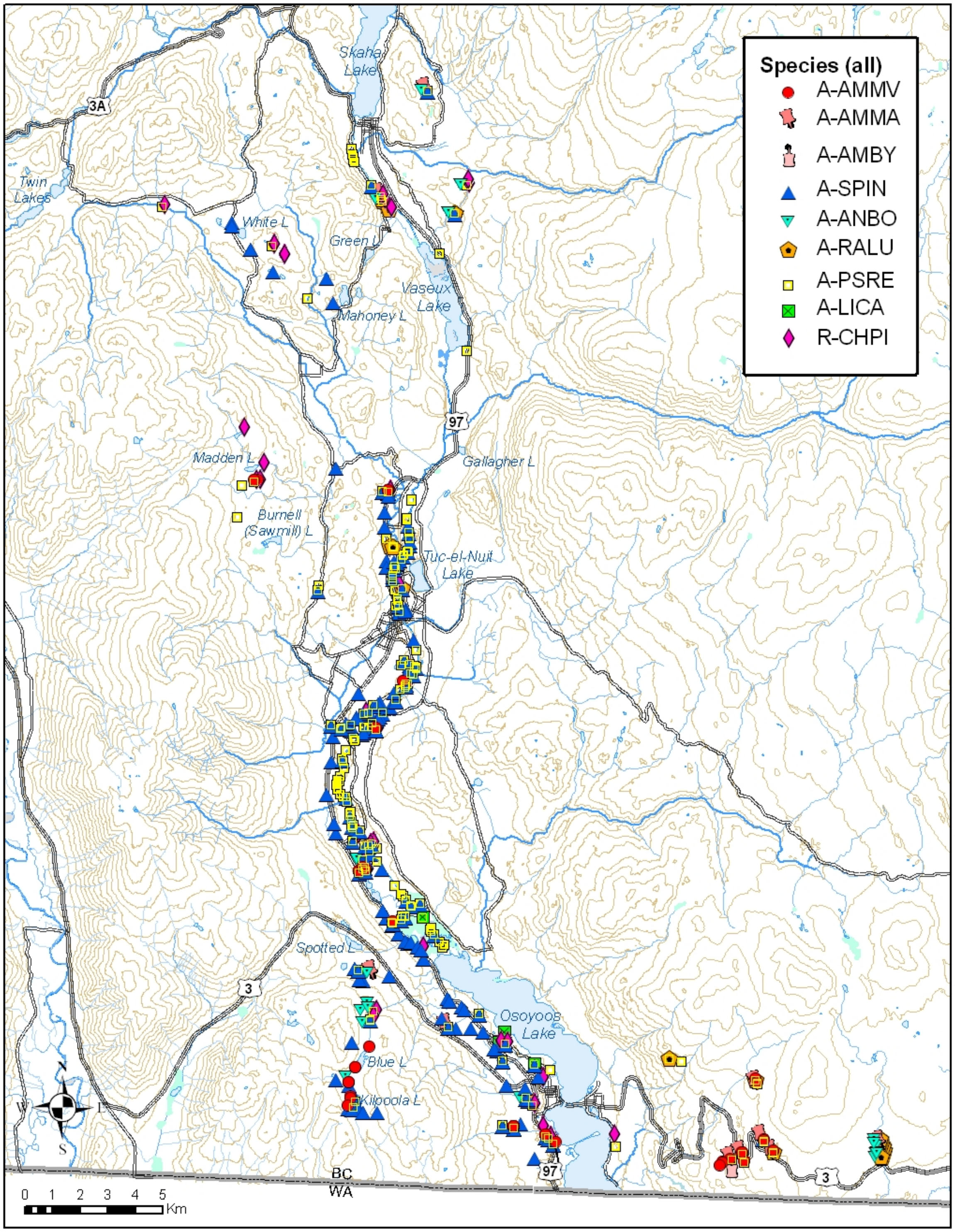

Figure 3.3. Total amphibian and turtle occurrence data ($N_{Records}$ = 2124), south Okanagan Valley, B.C., 2003 to 2006. Species codes: AMMA Long-toed salamander, AMMV Blotched tiger salamander, SPIN Great Basin spadefoot, ANBO Western toad, RALU Columbia spotted frog, PSRE Pacific chorus frog, LICA American bullfrog, PATU Western painted turtle, AMBY Unknown *Ambystoma* salamander species.

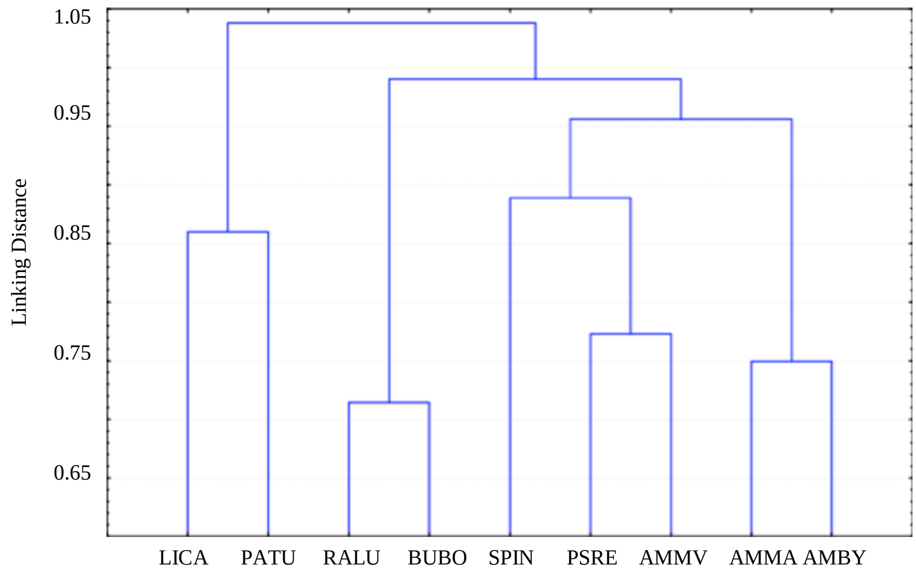

Figure 3.4. Tree diagram depicting likelihood of species coexisting in wetlands, south Okanagan Valley, B.C., 2003 to 2006. Species codes: AMMA Long-toed salamander, AMMV Blotched tiger salamander, SPIN Great Basin spadefoot, BUBO Western toad, RALU Columbia spotted frog, PSRE Pacific chorus frog, LICA American bullfrog, PATU Western painted turtle, AMBY Unknown *ambystoma* salamander species.

3.3.3.1 Species richness

Wetlands in the study area had low species richness, with greater than two-thirds of sites having less than two species detected annually (Table 3.6; Fig. 3.5).

Table 3.6. Summary of species richness (0 to 5 species detected) and the total number of corresponding wetland sites, south Okanagan, B.C., 2003 to 2006.

		Number species observed at a site					
Year	Number sites surveyed	Zero	One	Two	Three	Four	Five
2003	26	1	6	11	7	1	0
2004	61	16	14	16	13	2	0
2005	63	13	16	17	14	2	1
2006	73	7	18	29	17	2	0
Total number observations		37	54	73	51	7	1

<u>Land-use main classes:</u>

Species richness of wetlands significantly varied among main land-use classes (ANOVA: $F_{(3, 231)} = 6.37$, $p = 0.0003$). Species richness was significantly higher at agricultural sites compared to grazing sites (Tukey HSD: MSE = 1.13, df = 231, p = 0.05) and miscellaneous anthropogenic sites (Tukey HSD: MSE = 1.13, df = 231, p = 0.0002), but similar to reference sites (Tukey HSD: MSE = 1.13, df = 231, p = 0.45) (Fig. 3.6).

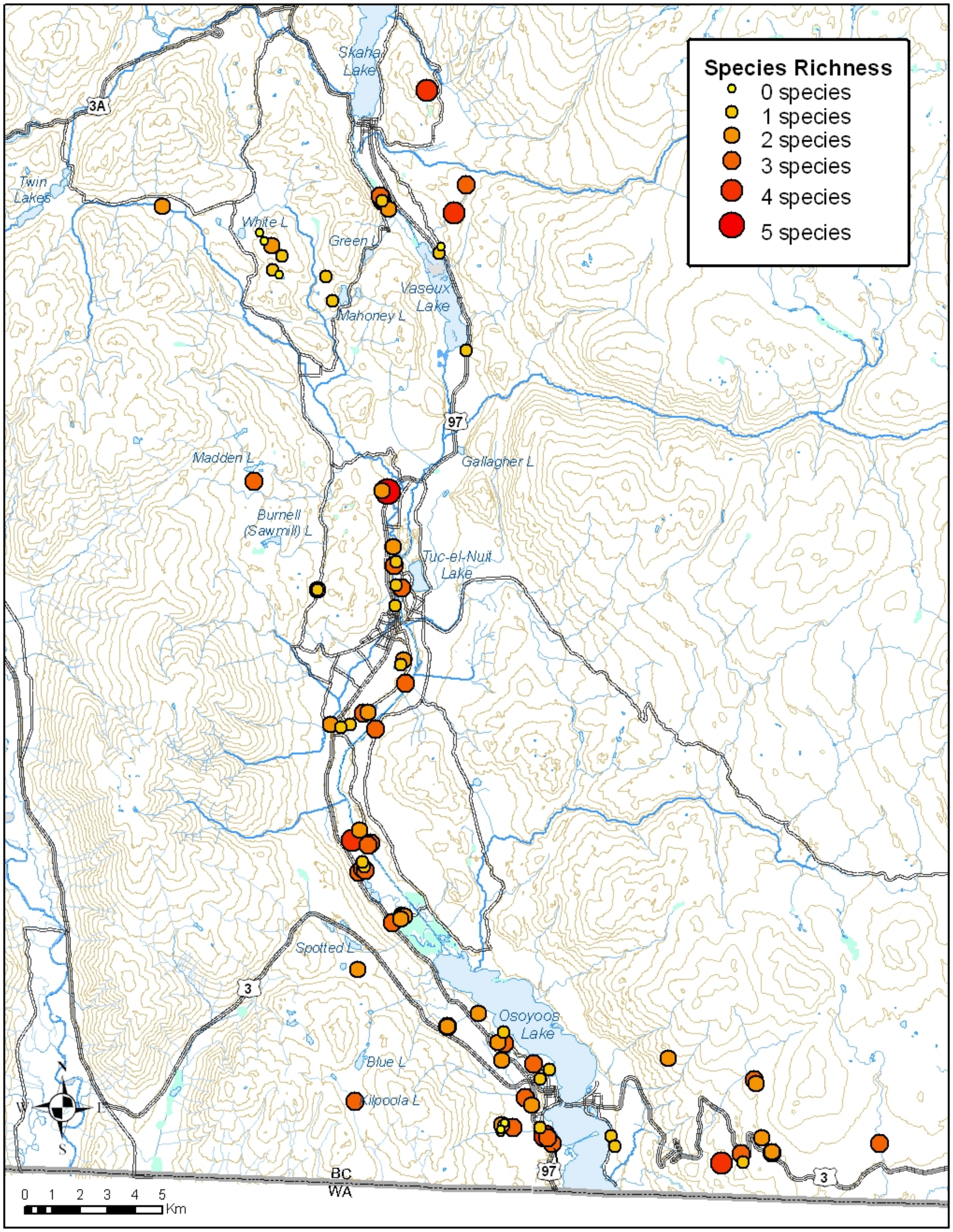

Figure 3.5. Discrete wetland species richness ranged from zero to a single site possessing five species of amphibians and one turtle species (as indicated by increasing circle size and colour hue), south Okanagan Valley, B.C., 2003 to 2006. Species included in analysis: Long-toed salamander, Blotched tiger salamander, Great Basin spadefoot, Western toad, Columbia spotted frog, Pacific chorus frog, Western painted turtle.

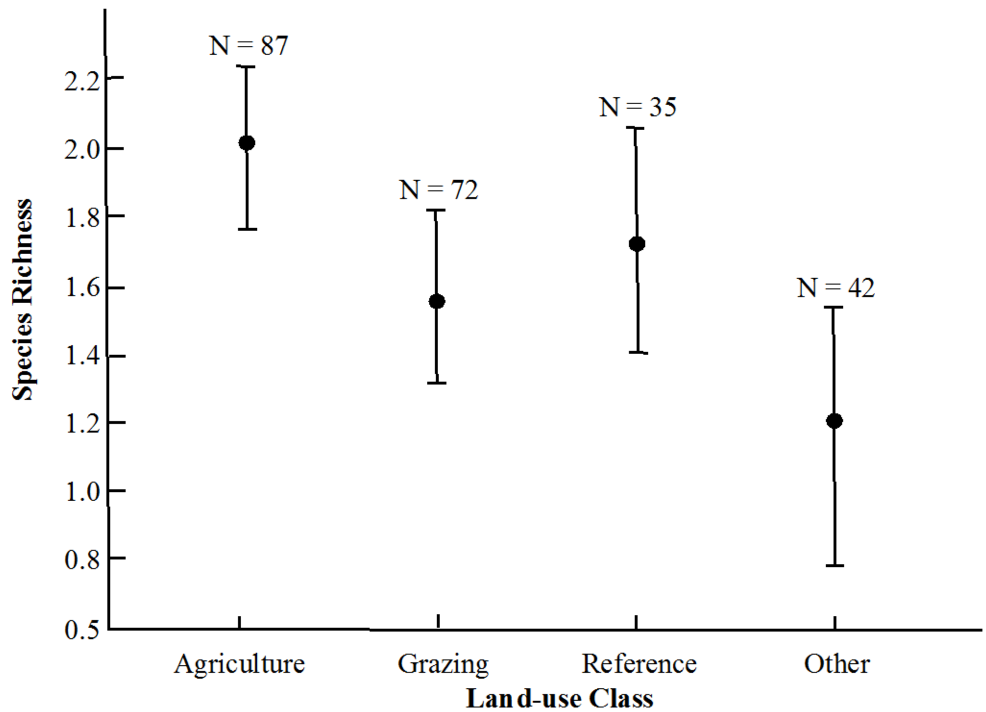

Figure 3.6. Species richness significantly varied among wetland main land-use classes (ANOVA), south Okanagan B.C., 2003 to 2006. Species richness was significantly higher at agricultural sites compared to grazing sites and other miscellaneous anthropomorphic, but similar to reference sites. Where N represents the total number of samples per site in the analysis. Bars denote 95% confidence intervals.

<u>Land-use sub-classes:</u>

Species richness significantly varied among wetland sites when assessing the land-use sub-classes (ANOVA: $F_{(7, 227)} = 4.73$, $p = 0.001$) (Fig. 3.7). Species richness was highest at conventional and organic agricultural sub-class sites. Organic sites were significantly higher in species richness compared to residential sites (Tukey HSD: MSE = 1.09, df = 227, p = 0.05) and protected grazing sites (Tukey HSD: MSE = 1.09, df = 227, p = 0.03). Conventional sites were significantly higher in species richness compared to residential sites (Tukey HSD: MSE = 1.09, df = 227, p = 0.003) and protected grazing sites (Tukey HSD: MSE = 1.09, df = 227, p = 0.001). Unprotected grazing sites had significantly higher species richness compared to protected grazing sites (Tukey HSD: MSE = 1.09, df = 227, p = 0.04).

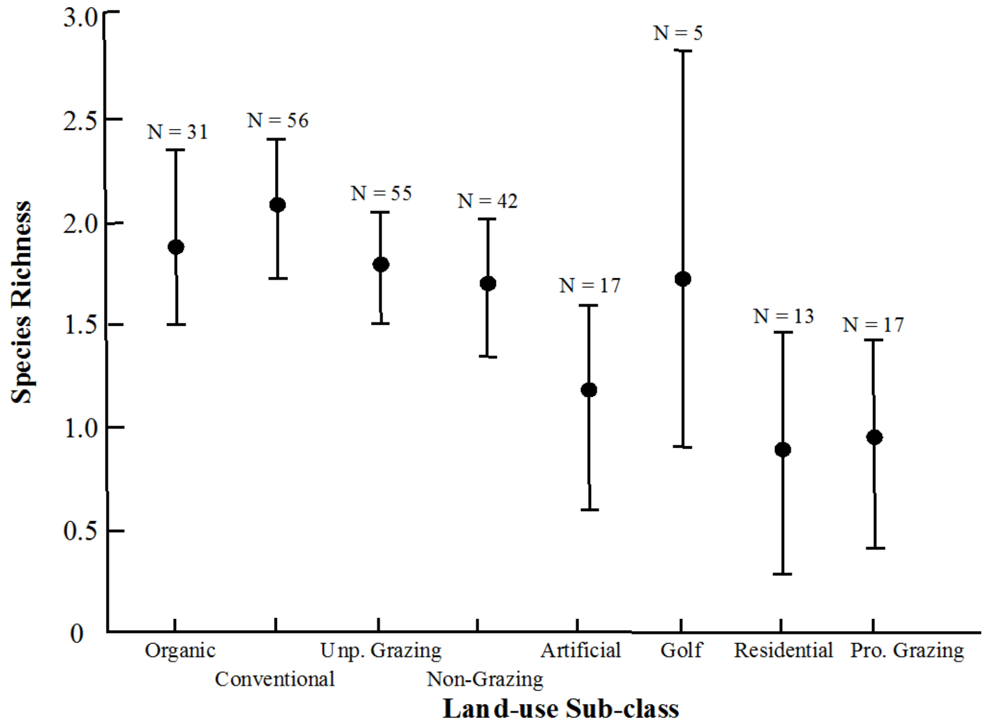

Figure 3.7. Species richness significantly varied among wetland land-use sub-classes (ANOVA), south Okanagan Valley, B.C., 2003 to 2006. Organic and conventional agricultural sites were significantly higher in species richness compared to residential sites and protected grazing sites. Unprotected (Unp.) grazing sites and non-grazing sites had significantly higher species richness compared to protected (Pro.) grazing sites. Where N represents the total number of samples per site in the analysis. Bars denote 95% confidence intervals.

3.3.4 Presence of fish

When using the monitoring dataset (Fig. 3.8) species richness was not significantly different among sites with the presence of fish (N = 53 records) compared to vacant sites (N = 164 records) (GLZ: Wald χ^2 (1) = 1.41, p = 0.24) (Appendix 3.7A). However, when species richness was compared to wetland sites with known early life stages, vacant wetland sites had significantly higher species richness compared to sites with invasive non-native fish (GLZ: Wald χ^2 (1) = 11.91, p = 0.001) (Fig. 3.9). No differences were observed among land-use sub-class (GLZ: Wald χ^2 (3) = 1.82, p = 0.61) (Appendix 3.7B) or between elevation sub-classes (GLZ: Wald χ^2 (1) = 1.37, p = 0.24) (Appendix 3.7C).

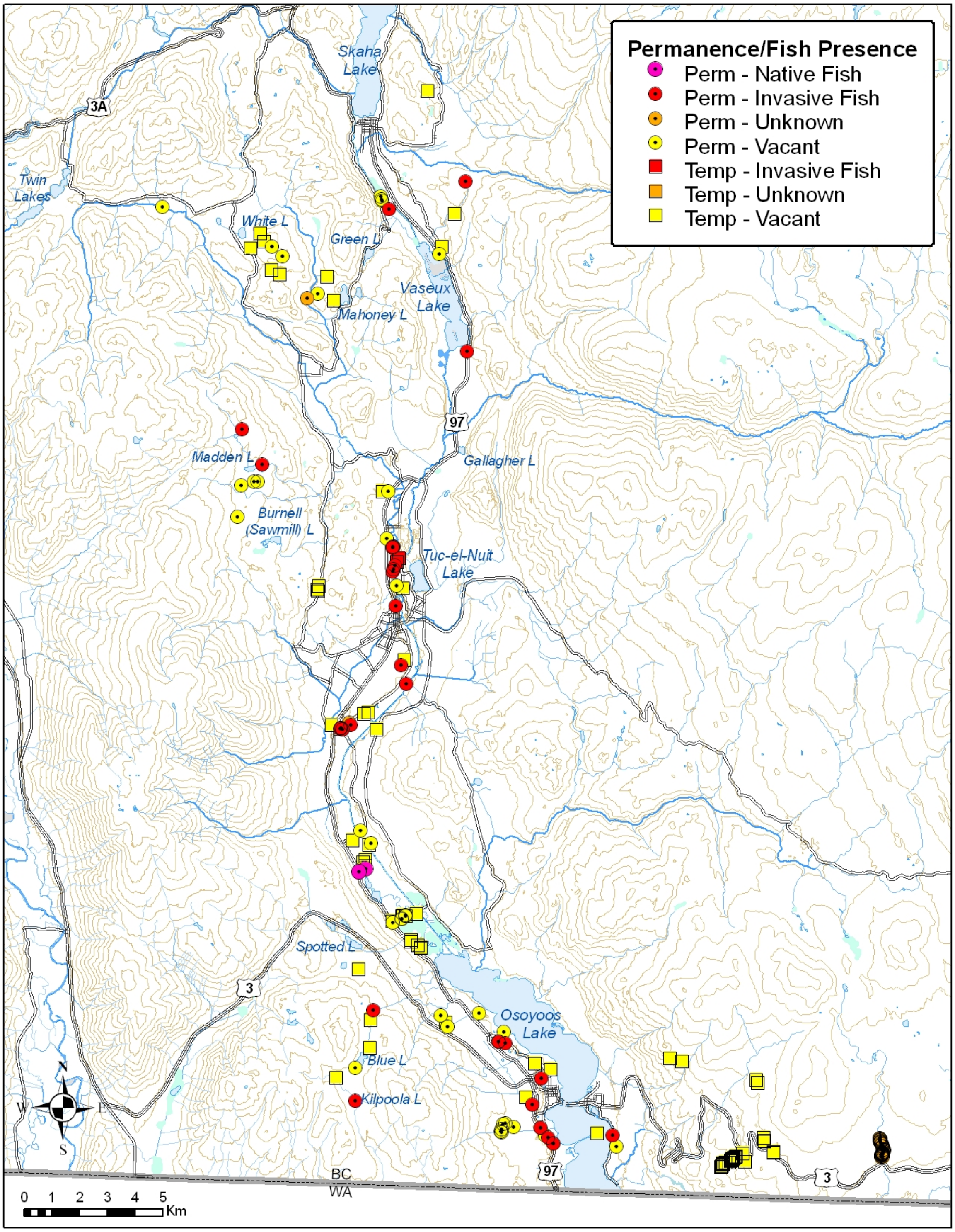

Figure 3.8. Permanent (Perm circle: N = 61 sites) and temporary (Temp squares: N = 53 sites) wetlands were categorized as being vacant (N = 77), non-native invasive fish species (N = 29), unknown (N = 6), or the detection of native fish (N = 2).

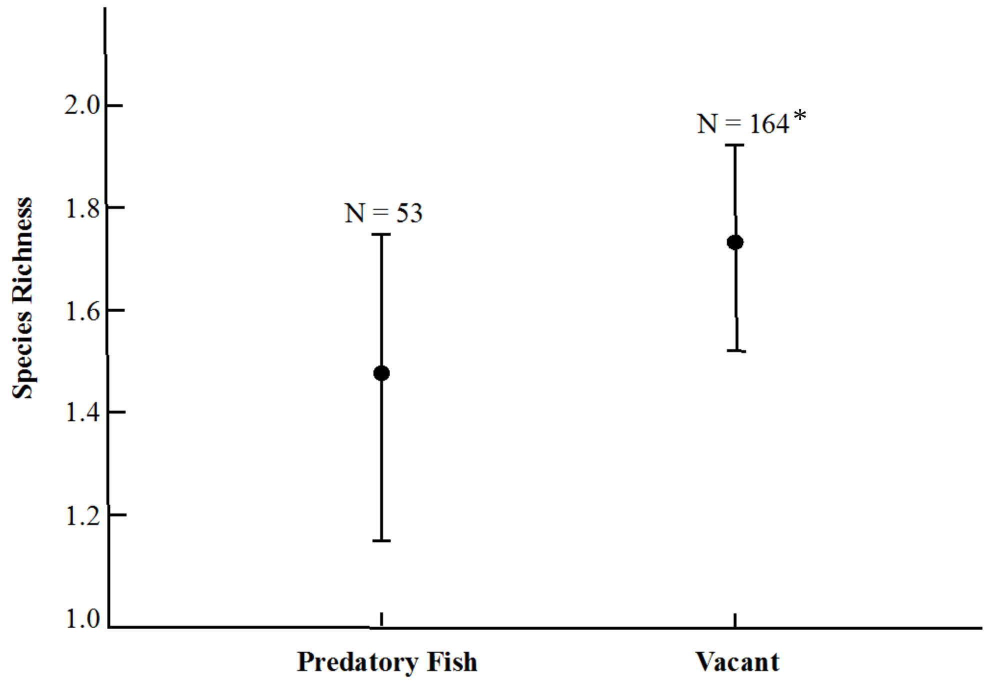

Figure 3.9. Species richness was significantly higher in wetland sites with no fish compared to sites with non-native invasive fish species, (GLZ: Wald χ^2 (1) = 11.91, p = 0.001*) south Okanagan Valley, B.C., 2003 to 2006. Where N represents the total number of samples per site in the analysis. Bars denote 84% confidence intervals.

3.3.5 Auditory call counts

Across all years, Pacific chorus frog calling was most frequently detected as overlapping individuals (index 2), followed by full choruses (index 3), and then individual calls (index 1). (Table 3.7; Appendix 3.8A). Whereas, Great Basin spadefoot auditory calling was most frequently detected as individuals (index 1) or a few overlapping individuals (index 2) (Table 3.7; Appendix 3.8B). The highest number of Great Basin spadefoot full choruses (index 3) was detected at the river channel at non-discrete sites suggesting greater habitat selection for this species for areas with ephemeral wetlands.

Table 3.7. Number of auditory sites surveyed annually and corresponding calling index (0 to 3) for Pacific chorus frog and Great Basin spadefoot, south Okanagan Valley, B.C., 2003 to 2006.

Year	Number sites Surveyed	Species	Calling Index			
			0	1	2	3
2003	27	Pacific chorus frog	5	5	5	12
		Great Basin spadefoot	12	7	8	0
2004	28	Pacific chorus frog	15	9	13	11
		Great Basin spadefoot	26	13	5	4
2005	81	Pacific chorus frog	29	11	27	14
		Great Basin spadefoot	60	16	3	2
2006	183*	Pacific chorus frog	41	24	96	22
		Great Basin spadefoot	113	22	37	11

*118 non-discrete sites surveyed.

3.3.5.1 Great Basin spadefoot calling index:

Great Basin spadefoot calling index varied significantly among main land-use classes (GLZ: Wald χ^2 (4) = 11.39, p = 0.02), with reference and river channel sites having significantly higher calling indexes than agricultural sites (Fig. 3.10). Similarly, Great Basin spadefoot calling index varied significantly among land-use sub-classes (GLZ: Wald χ^2 (4) = 15.17, p = 0.004), with non-grazing, and river channel sites significantly higher compared to conventional and organic agricultural sites (Fig. 3.11). Additionally, Great Basin spadefoot calling index significantly varied among elevation classes (GLZ: Wald χ^2 (1) = 5.26, p = 0.02, with calling index being higher at high elevation sites (GLZ: Wald χ^2 (1) = 5.77, p = 0.02). Great Basin spadefoot calling index did not differ significantly in the presence of fish (GLZ: Wald χ^2 (2) = 2.62, p = 0.27) or among water permanency classes (GLZ: Wald χ^2 (1) = 0.03, p = 0.86).

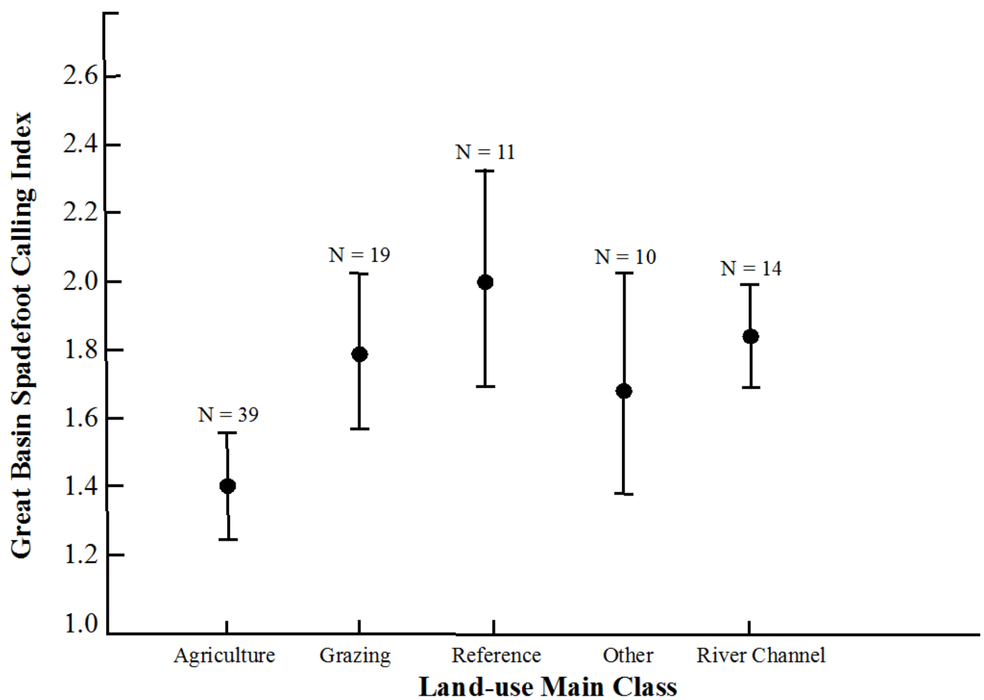

Figure 3.10. Great Basin spadefoot calling index varied significantly among main land-use classes (GLZ: Wald χ^2 (4) = 11.39, p = 0.02), with reference and river channel sites having significantly higher calling indexes than agricultural sites, south Okanagan Valley, B.C., 2003 to 2006. Grazing and 'other' sites were not significantly different from other sites. Where N represents the total number of samples per site in the analysis. Bars denote 84% confidence intervals.

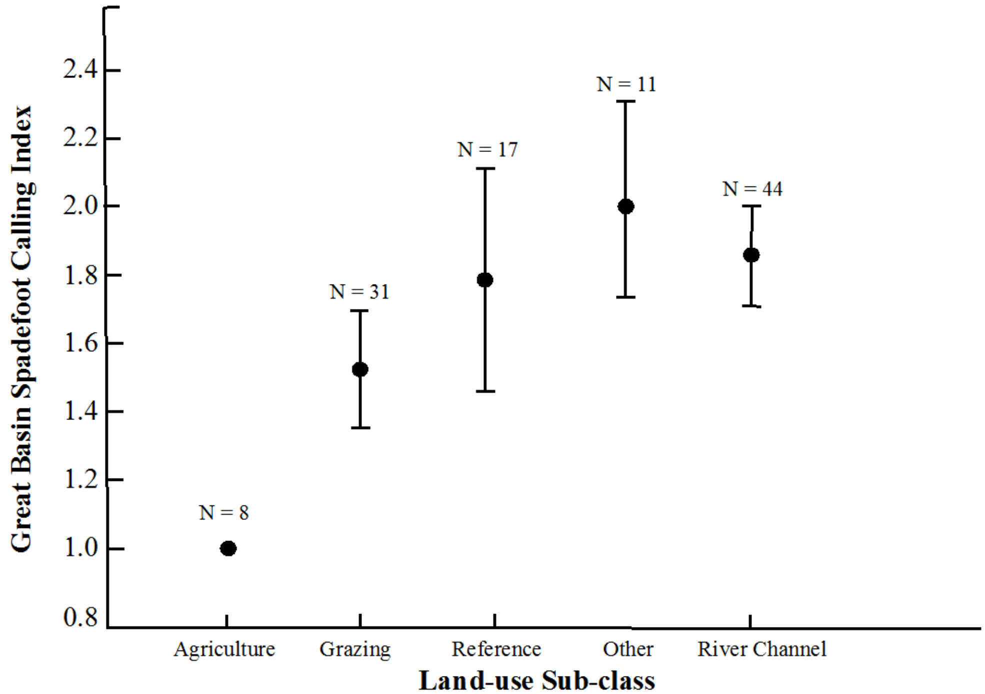

Figure 3.11. Great Basin spadefoot calling index varied significantly among land-use sub-classes with non-grazing, river channel, and unprotected grazing sites significantly higher compared to conventional and organic agricultural sites, south Okanagan Valley, B.C., 2003 to 2006. Where N represents the total number of samples per site in the analysis. Bars denote 84% confidence intervals.

3.3.5.2 Pacific chorus frog calling index:

Pacific chorus frog calling index did not differ significantly among main land-use class (GLZ: Wald χ^2 (4) = 7.06, p = 0.13), sub-classes (GLZ: Wald χ^2 (4) = 1.22, p = 0.87), elevation class (GLZ: Wald χ^2 (1) = 0.05, p = 0.82), or water permanency class (GLZ: Wald χ^2 (2) = 0.82, p = 0.37). However, Pacific chorus frog calling index varied significantly with the presence of fish (GLZ: Wald χ^2 (2) = 6.10, p = 0.04) and was lowest in the presence of invasive fish species (GLZ: Wald χ^2 (2) = 5.89, p = 0.05).

3.3.6 Relative density of early life stages

The relative density of early life stages among species was assessed at 64 sites, however due to small sample sizes statistical analysis was limited to Pacific chorus frogs, Great Basin spadefoots, and Western painted turtles. The most frequent density of early life stages observed among all species was very low (0 to 9 individuals) and occurred on 96 occasions (Fig. 3.12 Table 3.8). The Pacific chorus frog, a widely ranging species detected as egg or tadpole at 43 sites, was observed on only three occasions at high densities (> 1000 individuals; Appendix 3.9A). Similarly, Great Basin spadefoots were detected at 27 sites and observed on seven occasions at high densities (>1000 individuals; Appendix 3.9B) At low elevations early life stages were detected more frequently in permanent wetlands compared to temporary ones, whereas the reverse trend was observed at high elevations. According to land-use, non-grazing and unprotected grazing sites (N = 7 species each) had the largest number of early life stages observed by species, whereas anthropogenic sites (golf course N = 2 species, artificial pool N = 4 species, residential N = 7 species) were among the lowest (Appendix 3.10).

Table 3.8. Relative density of early life stages among sites by frequency category, elevation and water permanency classes (N = 64 wetland sites analyzed) south Okanagan Valley, B.C., 2003 to 2006. Species codes: AMMA Long-toed salamander, AMMV Blotched tiger salamander, SPIN Great Basin spadefoot, BUBO Western toad, RALU Columbia spotted frog, PSRE Pacific chorus frog, LICA American bullfrog, PATU Western painted turtle, A Amphibian, R Reptile.

	Number of Sites					Relative Density of Early Life Stages: Frequency of density category observed (2003 to 2006)			
		Low Elevation		High Elevation					
Species	Total* N_{sites}	Perm.	Temp.	Perm.	Temp.	Very low (1 to 9)	Low (10 to 99)	Medium (100 to 999)	High (≥ 1000)
A-PSRE	43	15	11	4	13	35	22	11	3
A-SPIN	27	7	4	1	5	15	14	11	7
R-PATU	18	11	3	4	0	18	0	0	0
A-AMMV	16	7	0	0	9	15	3	2	0
A-AMMA	9	0	2	1	6	8	6	0	0
A-RALU	6	1	0	1	4	1	5	0	2
A-BUBO	4	0	0	2	2	2	1	0	1
A-LICA**	2	2	0	0	0	2	1	1	1
Total						96	52	25	12

*Total number of sites where species observed; **American bullfrogs were being actively removed from the two conventional orchard sites (data not presented).

Relative density (RD) of early life stages among Great Basin spadefoots (GLZ: Wald χ^2 (3) = 2.69, p = 0.44), Pacific chorus frogs (GLZ: Wald χ^2 (3) = 2.69, p = 0.44), and Western painted turtles (GLZ: Wald χ^2 (2) = 3.13, p = 0.21) did not significantly differ within species among years. Similarly, the relative density of early life stages did not differ significantly among main land-use class for Great Basin spadefoots (GLZ: Wald χ^2 (3) = 5.33, p = 0.15), Pacific chorus frogs (GLZ: Wald χ^2 (3) = 1.55, p = 0.67), or Western painted turtles (GLZ: Wald χ^2 (3) = 5.45, p = 0.14). However, relative density of early life stages was significantly different among sub-classes for Pacific chorus frogs (GLZ: Wald χ^2 (7) = 17.63, p = 0.01) with conventional orchards having the highest relative density (RD 1.59: 1.27 – 1.91, 84% CI) (Fig. 3.13, Appendix 3.11A) and protected grazing sites the lowest (RD 0.17: 0.12 – 0.44, 84% CI). Similarly, Western painted turtles (GLZ: Wald χ^2 (5) = 17.41, p = 0.004; due to low sample sizes golf course and artificial sub-classes were dropped from analysis) density was highest at protected grazing sites (RD 0.20: 8.25 - 1.86, 84% CI) Appendix 3.11B. No differences among mean relative densities were found among subclass sites for Great Basin spadefoots (GLZ: Wald χ^2 (4) = 8.65, p = 0.071; due to low sample sizes golf course, protected and residential grazing sub classes were dropped from analysis) (Appendix 3.11C). However the highest relative density of early life stages for a species was observed among Great Basin spadefoots in non-grazing sites (Appendix 3.11D).

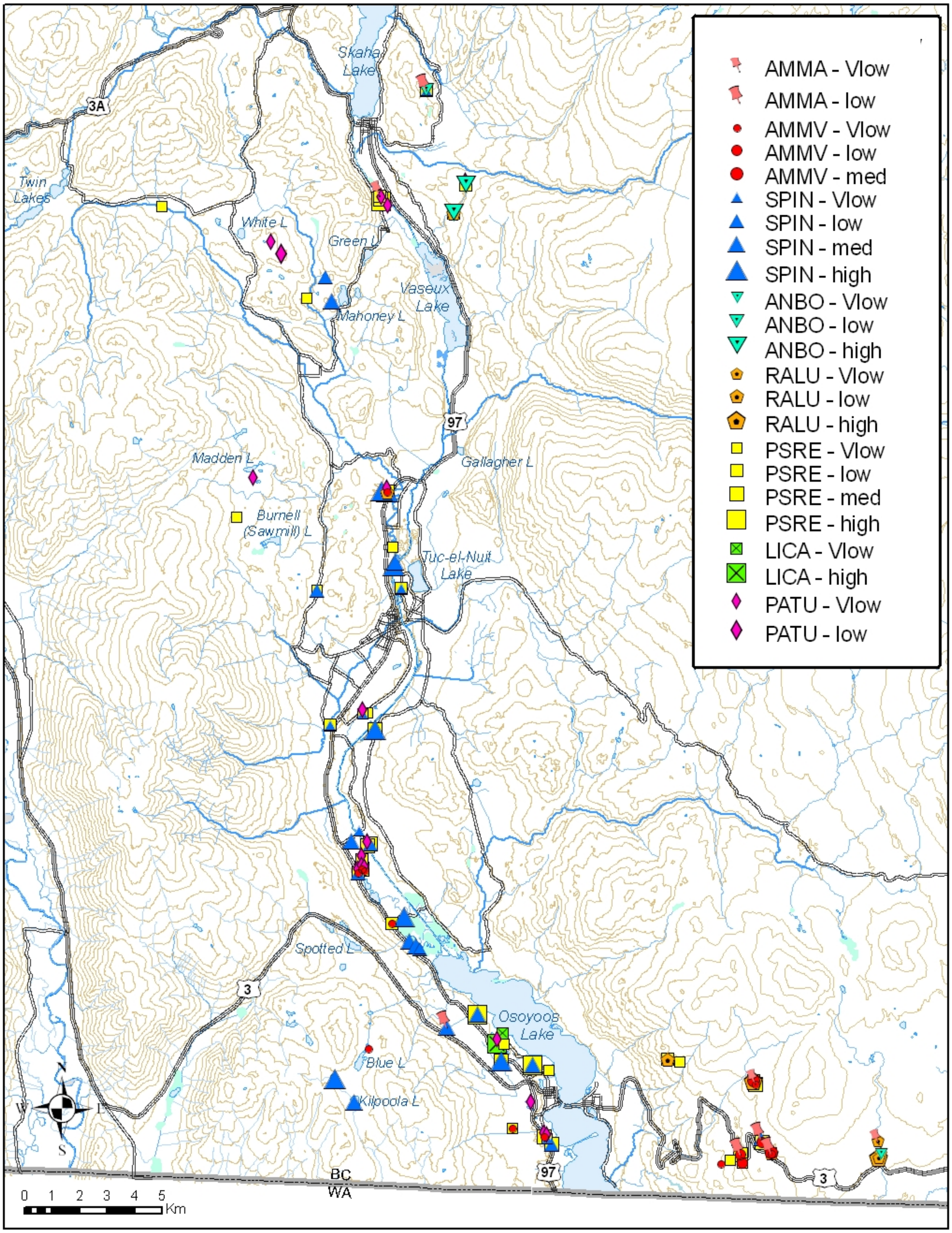

Figure 3.12. Species relative density of early life stages observed at 64 discrete wetland sites, south Okanagan Valley, B.C., 2003 to 2006. The number of early life stage individuals was categorized as very low (Vlow) = 1 to 9, low = 10 to 99, medium (med) = 100 to 999 or high ≥ 1000 individuals). Species codes: AMMA Long-toed salamander, AMMV Blotched tiger salamander, SPIN Great Basin spadefoot, BUBO Western toad, RALU Columbia spotted frog, PSRE Pacific chorus frog, LICA American bullfrog, PATU Western painted turtle.

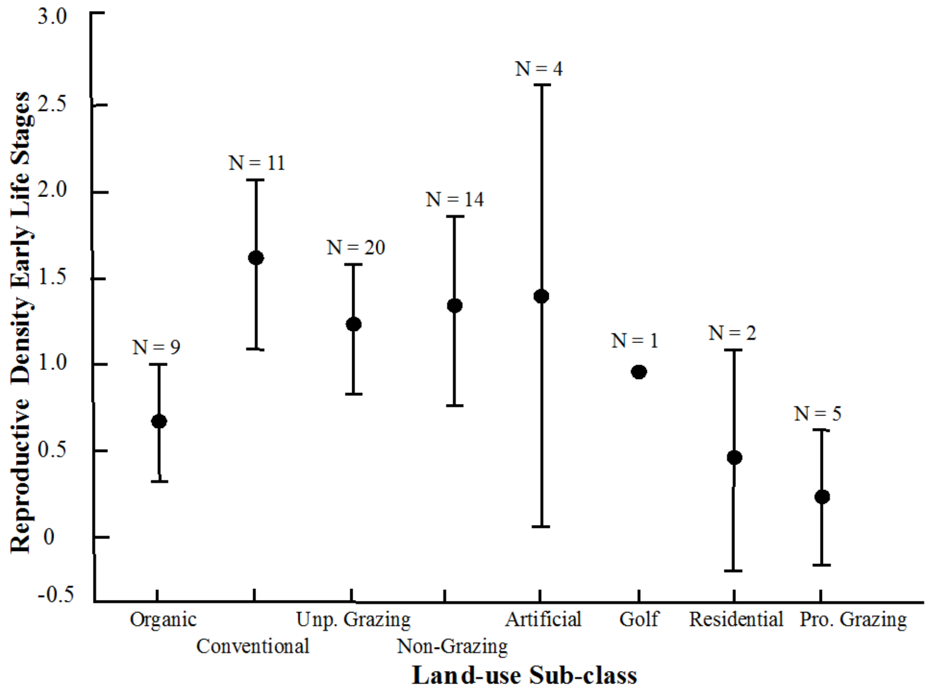

Figure 3.13. Relative density of early life stages of Pacific chorus frogs among sub classes, where non-overlapping bars indicate significant differences in relative density. Early life stages of species in conventional orchards, unprotected grazing, and non-grazing sites are significantly higher than in protected grazing sites. Where N represents the total number of samples per site in the analysis. Bars denote 84% confidence intervals.

Relative density of early life stages did not significantly differ between elevation classes for Pacific chorus frogs (GLZ: Wald χ^2 (1) = 0.16, p = 0.70) nor Great Basin spadefoots (GLZ: Wald χ^2 (1) = 0.96, p = 0.33), but were significantly higher among the high elevation sites for Western painted turtles (GLZ: Wald χ^2 (1) = 9.82, p = 0.002). Mean relative density of early life stages did not significantly differ between water permanency classes for Pacific chorus frogs (GLZ: Wald χ^2 (1) = 1.51, p = 0.22) nor Western painted turtles (GLZ: Wald χ^2 (1) = 1.65, p = 0.20), but was significantly higher in temporary wetlands for Great Basin spadefoots GLZ: Wald χ^2 (1) = 7.13, p = 0.01). Pacific chorus frogs had significantly higher density of early life stages in ponds vacant of fish (GLZ: Wald χ^2 (1) = 8.27, p = 0.02), compared to Great Basin spadefoots or Western painted turtles (GLZ: Wald χ^2 (1) = 0.05, p = 0.82; GLZ: Wald χ^2 (1) = 0.001, p = 0.92)

3.4 DISCUSSION

Our hypothesis was supported in that species richness, distribution, and relative density of herpetofauna among 108 wetlands surveyed during 2003 to 2006 in the south Okanagan Valley show significant differences among sites as defined by their land-use characteristics. Several factors and trends emerge when assessing the survival of wetland dependent species that require a complex of wetlands. Particularly at low elevations, the south Okanagan Valley amphibian and turtle populations are using relatively few breeding ponds in a fragmented terrestrial landscape. Amphibian and reptile habitat-use patterns appear to indicate that they are not selecting for the most pristine habitats given they readily use conventional agricultural sites that had experienced the highest number of potential stressors. Regardless of habitat subclass, the species richness and density of species was low in almost all sites, and some trends remain unexplained.

This is the first survey of the species richness, distribution, and relative density of amphibians and the western painted turtle in the south Okanagan Valley. Although some previous inventory of amphibians and reptiles exists in the south Okanagan Valley (B.C. Conservation Data Center), the absence of long-term regional species database prior to significant development, makes establishing temporal or spatial evidence of population trends anecdotal. Similarly, since our analysis is correlative it is not possible to determine the mechanism driving our findings. However, several factors and trends emerge when assessing the survival of wetland dependent species that require a complex of wetlands in the Okanagan Valley that may be indicative of amphibian and reptile populations in other stressed habitats especially in the arid valleys of western North America.

Our analysis suggests that amphibian and reptile habitat-use patterns in the south Okanagan valley are based on the degree of habitat selectivity of the organism, meeting physiological needs, regardless of the presence of potential anthropogenic stressors we measured. Conventional agricultural sites had the highest potential stressor assessment, one of the highest species richness and some of the highest observed densities of early life stages. Though our hypothesis was supported, it is important to recognize that regardless of subclass the species richness and relative density of early life stages of all species was very low. In particular, both salamander species, Columbia spotted frog, and Western toad were often omitted from statistical analysis due to small sample sizes, and as such made interpretation of their population status or trends challenging but suggest number and distribution of these species are extremely low and fragmented in occurrence. Further, higher species richness trends among our unprotected sites are not easily explained, but may be a effect of surrounding natural landscape features in the higher elevations which were not assessed. The existing native amphibian and turtle species in the lower valley appear to represent remnant populations constrained by geographical and anthropogenic barriers. Consequently, species in the lower valley may have increased likelihood of annual wetland-level reproductive failure coupled with a low probability of species re-colonization due to isolation and environmental or anthropogenic stressors.

3.4.1 Landscape mosaic and potential anthropogenic stressors

Both site-specific parameters and landscape scale patterns of land-use appear to influence species occurrence and population dynamics (Hutchens and DePerno, 2009). Thus, it is important to understand landscape variation and human impact/modification in the management of wetlands and upland habitat mosaics for amphibian and turtle conservation. Further, both human dominated change and climate processes alter the abiotic and biotic components of habitats in ways that affect landscape suitability for species of conservation concern (Blaustein and Kiesecker, 2002; Saenz, et al., 2006). In support of our findings, mapping analyses of the Okanagan Valley provides habitat-based evidence that lowland populations of wildlife are becoming increasingly isolated and disconnected from their aquatic and terrestrial habitats and from the upper elevation populations (Lea, 2008). Few natural wetlands remain in the south Okanagan Valley and the habitat variability observed among wetlands is most likely a direct outcome of historic and ongoing wetland modification, including a history of agricultural irrigation and grazing water access using artificially constructed ponds. A wide range of variation among habitat parameters, coupled with low species occurrence, makes assessment of trends challenging and leaves individual contributions of parameters poorly quantifiable.

While general habitat descriptions are generally inadequate to predict presence and abundance of many species (Block and Morrison, 1998), several wetland characteristics we measured have been linked with amphibian and turtle population dynamics. Amphibian species occurrence has been positively correlated with pond depth (Jakob et al., 2003) and the absence of predatory fish (Ryan et al., 2014), while open surface water and some water chemistry parameters are negatively associated with the presence of some amphibian species (Jakob et al., 2003; Nyström et al., 2007). Our findings likewise support the importance of maintaining fishless ponds where native species have not adapted to fish predators and permanent wetlands are required breeding habitats for some species. Most notably, breeding and movement behaviour of the Great Basin spadefoot was observed in response to water management practices in the lowland valley. We found asynchronous amphibian breeding with irrigation and in some cases rapid draining or evapotranspiration of ponds resulting in complete reproductive failure. Asynchronous breeding has also been observed in Great Basin spadefoot using stream systems where there was no risk of pond drying (California: Morey et al., 2004), whereas synchronous breeding is observed consistently among other Spadefoot subspecies (Greenberg and Tanner 2004; Morey et al., 2004). A closer examination of local hydrological conditions and amphibian breeding success to metamorphosis in the south Okanagan is needed to predict population threats due to land-use water management and climate change interactions. Mitigation using existing irrigation infrastructure to manipulate water levels and habitat enhancement could be directed to reduce desiccation and thermal stress (Luke et al., 2011)

The occurrence of amphibians and turtles at agricultural sites, albeit at very low densities and very low species richness, suggests some tolerance to disturbance. The frequency of potential anthropogenic stressors around ponds on private land was substantial and extended throughout the south Okanagan Valley landscape with land-use practices. Small stressors on individual sites may also result in considerable cumulative effects on a landscape scale. The most cumulative anthropogenic effects were observed at ponds with agricultural encroachment, followed by urban and golf course sites. Practices still used, as observed by the author (S. Ashpole), but that appear to be on are on a regional decline due to local stewardship efforts (Okanagan Similkameen Stewardship Society; South Okanagan Similkameen Conservation Program) include excessive garbage dumping and burning around wetlands (2005 open air burning bylaw No. 2364, Regional District Okanagan Similkameen), which result in infilling and denuded or narrow buffers with weedy vegetation. In many cases the distance to agricultural crops appears to be either a response to tractor maneuverability or to maximize crop rows.

On a localized level, the predatory nature of the American bullfrog and fish species is also likely linked to the absence of amphibian species at those sites where they occur. A study examining effects of both predatory fish combined with American bullfrogs found additive negative effects on the occupancy of native amphibians (Preston et al., 2012). On a regional level it can be inferred that sites with non-native invasive species likely act as a population sink for colonization and hinder movement of species. Introduced non-native fish species are pervasive throughout the valley's watercourses and ponds. Fish stocking is connected to recreational fishing, aquaculture, ornamental, and mosquito control. More than 14 known fish species have been introduced locally since the 1900s (Rae, 2005). Small ponds are at increased danger of intentional stocking due to their proximity to humans and the perceived health risk, for example, Messaging by the Minister of Interior Health has been misinterpreted on some occasions, the statement "Install a pump in ornamental ponds or stock them with fish" (News Release, Apr. 2010. Interior Health) and "Pumps should be used in ornamental ponds to circulate water (thus drowning larvae), or ponds should be stocked with goldfish (which consume larvae)" (RDCO West Nile Virus Program, 2004 - 2008 BWP Consulting Inc., Page 7) was interpreted as direct advise to stock small ponds specifically with fish (pers. comm. with local landowners and NGOs).

The absence of native amphibian species at wetland sites with fish detection is not surprising given the probable ecological naivety of local Okanagan amphibian species. The most commonly observed fish species in the study are thought to depredate amphibian eggs and larvae and include the common carp (*Cyprinus carpio*), goldfish (*Carassius auratus;* Monello and Write, 2001), and bass sp. (*Micropterus* sp.). Further changes include evidence of alterations in community structure, trophic dynamics, reduced water quality and habitat degradation have been established with carp (*see* review Nieoczy and Kloskowski, 2014). The most frequently observed fish species in the study are thought to depredate amphibian eggs and larvae, including the common carp (*Cyprinus carpio*), goldfish (*Carassius auratus;* Monello and Write, 2001), and bass sp. (*Micropterus* sp.). Further impacts include evidence of alterations in community structure, trophic dynamics, reduced water quality and habitat degradation has been established with carp (*see* review Nieoczy and Kloskowski, 2014). Habitat enhancement measures are a priority to removing disturbances by implementing active management in partnership with agriculturalists and conservation based programs. Unfortunately the removal of fish logistically with community support is challenging and has only successfully occurred at two sites (*see* Chapter 4).

Terrestrially, sandy soils have been positively associated with both salamander (Block and Morrison, 1998) and European spadefoot (*Pelobates fuscus*) occurrence (Nyström et al., 2007). As would be anticipated for fossorial species, substrate and substrate modification can significantly affect a species defense mechanism to burrow, movement patterns across a landscape, and availability of refugia (Green-striped burrowing frog, *Cyclorana albogutta*: Booth, 2006). The high occurrence and relative density of early life stages of Great Basin spadefoots not surprisingly overlaps with the well drained sandy soils and irrigation ponds often located in agricultural sites. While the study suggests the availability of microhabitat suitability at agricultural sites, the permeability of upland habitat in conventional agricultural sites is significantly more compact compared to organic and reference sites with a trend of increasing impermeability with distance from the wetland (*see* Chapter 4) Additionally, frequent tilling in agricultural systems has been implicated in increased mortality of *Pelobates fuscus* (Nyström et al., 2007) and would likely have similar consequences for other burrowing species. Observations of tilling activity in the Okanagan Valley were variable, with weed management and fields used for ground crops or transitioning to vineyard the most frequent.

Highway expansion in the south Okanagan continues to bisect breeding sites in the lower floodplain from upland terrestrial habitat resulting in significant amphibian road mortality

(Crosby 2014). The population level effect of road mortality for most species has yet to be quantified, and the response in declining species occurrence may occur gradually due to a time lag between changes in urban landform and road traffic intensity - this was the case in Sweden (Löfvenhaft et al., 2004). All of the low elevation ponds in the south Okanagan Valley are located near road systems and within the distance required for amphibian species movement within a metapopulation scale for landscape level conservation (Semlitsch, 2008) and where noise disturbance can alter anuran-calling behaviour (Cunnington and Fahrig, 2013). The close proximity of ponds to roads (Nyström et al., 2007) and varying response of different species to traffic intensity (Mazerolle, 2003; Pellet, et al., 2004) has been implicated as a significant demographic force resulting in population level declines of amphibian species that make overland movements to breeding grounds. During overland nesting movements of female turtles the animals are killed disproportionately (Gibbs and Steen, 2005) and slight declines in adult turtle survival can lead to large population collapses (Congdon et al., 1993).

As the residential and seasonal population and corresponding development expand in the south Okanagan the negative effects on urban and peri-urban-agricultural wetlands increases. The residential and urban golf course ponds in the south Okanagan Valley commonly create a sink habitat wherein increased hydroperiod of the ponds combined with presence of predatory fish that has been linked to lower species richness (Rubbo and Kiesecker, 2004) and variable species resilience (e.g. European tree frog *Hyla arborea*; Pelet et al., 2004). Habitat suitability modeling of urban wetlands and most golf course ponds in the south Okanagan Valley suggests they can act as movement corridors and suitable colonization sites for predatory American bullfrogs (Lukey et al., 2012). However, golf courses managed and designed with biodiversity and ecosystem service goals have resulted in increased species richness (Colding and Folk 2009). The only species detected at residential sites was the Western painted turtle and one American bullfrog. No species were detected at one of the golf course site ponds. Whereas, the fishless golf course pond site had breeding Pacific chorus frogs, Great Basin spadefoots, and Blotched tiger salamanders present. While this particular fishless golf course pond is adjacent to natural habitat features, it is exposed to nutrient additions, isolated from other wetlands by more than 1.5 km and bisected by a main highway. The upland terrestrial habitat likely contributes to site persistence and possible emigration from the upper benches, however within the lower valley the metapopulation dynamics are likely challenging with an adjacent pond having non-native fish species and the other completely drying in some years.

3.4.2 High elevation stressors: livestock grazing and off-road vehicles

Wetlands at higher elevations were generally associated with grazing stress or the introduction of predatory sport fish. Livestock grazing is the dominant land-use among the high elevation sites and plays an important role in maintaining the suitability of ephemeral hydrological conditions for California tiger salamander (*Ambtstoma californiense*) reproduction (Pyke and Marty, 2005). Conversely, grazing has been associated with changes in frog communities and decreased habitat condition (Jansen and Healey, 2003), possibly due to altered littoral habitat complexity. A universal assumption about the impact of cattle grazing management, habitat alteration, and amphibian responses is not easily deciphered by the literature. Reduced habitat complexity has been implicated as the reason ranid species have shown reduced abundance and body size with grazing activity, whereas bufonid species have shown the opposite effect (Burton et al., 2009). Establishing a habitat condition index (*see* Burton et al., 2009) in the south Okanagan Valley may help to clarify why the unprotected grazing sites had significantly higher species richness compared to sites with restricted livestock access. Fencing remains a regional priority to facilitate the protection of watercourses from livestock through exclusion fencing and controlled water access. Since 2001, approximately 64,109 km of fencing around 835 hectares has been installed in the south Okanagan at a cost of about $40,000 year. The inhibitive cost of fencing and challenge to uphold grazing regimes has likely slowed the progress of local stewardship groups (pers. comm. M. Sarell and A. Haney, Aug. 2014).

3.4.3 Water chemistry

The input of nutrients into aquatic systems may be widespread throughout the valley with multiple sources, including agricultural practices, livestock run-off, golf course turf fertilization, and urban additions. The wetlands sites surveyed in the study are comparatively small and may be particularly sensitive to nutrient and contaminant additions. The differences in water quality parameters between early and late sampling collection suggest repeat water sampling timed with land-use practices would be useful in future to develop a seasonal profile and more accurate risk assessment. Water quality parameters that exceeded the Canadian Water Quality Guidelines (CWQG) for the protection of aquatic life (Canadian Council of Ministers of the Environment. 2007) included levels of chloride, fluoride, nitrite, pH, and ammonium. The nitrite levels were highest among golf courses (8.4 mg NO_2^-/L.) and organic agricultural sites (2.7 mg NO_2^-/L.) where experimental concentrations of nitrite, ranging from 0.22 to 7.0 mg NO_2^-/L, can negatively affect Pacific chorus frogs and Oregon spotted frog (*Rana pretiosa*) activity eventually leading to death (Marco et al., 1999). Our total nitrogen levels average exceeded 2.1 mg/L, similar to field levels reported for total nitrogen concentrations, which were an important factor separating ponds with European spadefoot toad (*Pelobates fuscus*) reproduction from ponds without (Nyström et al., 2007). Increased algae productivity due to higher concentrations of nitrogen and phosphorus may contribute to the presence of increased life stages at some agricultural sites, but the relationship is not apparent in the protected grazing sites. In a concurrent study, water chemistry parameters such as ammonia, nitrates, sulfate, and phosphorus correlated with decreasing hatching success of native amphibian species in the south Okanagan Valley (Bishop et al., 2010). Not surprisingly the concentration levels were similar in the present study with the exception of higher ammonia levels.

3.4.4 Variation in results by species surveys

Each survey method may introduce bias and provide variable detectability (for estimation techniques *see* Jung et al., 2002). In particular our analysis did not accommodated bias that may occur when a sites is surveyed in entirety versus a sites where a restricted area was surveyed possibly excluding suitable microhabitats. To reduce sampling bias, we used multiple, common amphibian survey methods including both daytime and nighttime searching (Shaffer and Juterbock, 1994). While visual encounters and dip netting can underestimate abundance and population size with increasing surface area (Jung et al., 2002) and is subject to bias among the larger ponds with restricted access. To determine more accurate amphibian population numbers mark-recapture studies could be incorporated as select sites, although the effort is time-consuming and the level of invasiveness increases considerably (Jung et al., 2002). While the timing of monitoring activities was carefully managed to increase the probability of species detectability, the findings may underestimate the occurrence of some species. Future studies in the south Okanagan should consider additional techniques to assess the rarely recorded Blotched tiger salamander (e.g. pitfall) and Western painted turtle (e.g. hoop traps versus opportunistic) species. The timing of Blotched tiger salamander movements, egg laying, or presence of early larvae were not determined due to the lack of data (e.g. die off, introduction of goldfish, Ashpole et al., 2011) or survey timing (e.g. metamorphic emergence late July, pers. comm. O. Dyer). Monitoring timing is critical to record desert-adapted amphibians, such as the Great Basin spadefoot who can have a narrow window of calling, explosive breeding, and a rapid rate of development. While many amphibian species depend on rainfall events to initiate breeding behaviour (Saenz et al., 2006), agricultural irrigation has been observed to facilitate Great Basin spadefoot movements, pond filling, and subsequent asynchronous breeding (Ashpole et al., 2014).

3.4.5 Species richness and relative density

Although the number of amphibian species occurring in the south Okanagan Valley is considered relatively high for the arid interior of B.C. (Matsuda et al., 2006), of all 108 sites surveyed over the years, only three specific sites consistently had 4 or 5 breeding species detected. The common landscape feature of these sites is adjacent natural habitat without road bisection. An important factor in identifying priority ponds for conservation action is to determine which sites on private lands potentially contribute to high species richness and provide source population within the valley. Our survey effort and results suggest that ponds on private land with any native amphibian or turtle species with observed early life stages should be considered a part of a priority assessment to direct stewardship efforts. Sites at risk with very low detections of early life stages and that appear to have high annual variability in breeding success include lowland permanent ponds with Blotched tiger salamanders and Western painted turtles and ephemeral ponds with Great Basin spadefoots. However, the Great Basin spadefoots use of multiple habitat types and wide ranging distribution may provide greater opportunity for recovery actions as indicated by the increase use of permanent wetlands in the lower valley compared to high elevation sites.

3.4.6 Species assemblages

The south Okanagan wetland mosaic provides a diversity of permanent and temporary habitats used by amphibian and Western painted turtle species. The extent of individual species range varies spatial from large geographical areas (e.g. Pacific chorus frog) to highly restrictive narrow ranges (e.g. Blotched tiger salamander). Depending on hydrological conditions or the time of year, species may use and move between wetlands for various purposes. Species that are long-lived tend to tolerate some years with reproductive failure (e.g. Marbled salamander, *Ambystoma opacum;* Taylor et al., 2006), whereas other species are explosive breeders with dramatically fluctuating populations. The level of terrestrial dependence and length of time spent at wetlands is also highly variable among adults and the larval development of species.

The cluster analysis grouped species that are likely to co-exist together, and the composition of fauna can be explained by habitat selection by species. Our cluster analysis tree combines species together according to their dependency on permanent waters. American bullfrogs and Western painted turtles were linked in the analysis and observed together likely due to their highly aquatic nature, ability to coexistence with fish (Bury and Whelan, 1984), and preference for over wintering in permanent waters (Wilbur and Collins, 1973). Additionally, water permanency is essential in the multi-year larval development of American bullfrogs (Wilbur and Collins, 1973). Similarly, tiger salamanders species may take multiple years to reach sexual maturity and even remain aquatic in some cases (Whiteman et al., 2012). In the lower valley, Blotched tiger salamander sites were within close proximity each other and only occupied permanent wetlands absent of non-native invasive fish species (Ashpole et al., 2011). All the remaining amphibian species can achieve larval development within a season. The Pacific chorus frog was closely branched with the Blotched tiger salamander which might be due to this species diverse use of habitat types, from ephemeral to permanent. Columbia spotted frogs and Western toads were linked in the cluster analysis and most frequently observed in our study using larger flowing wetland systems at higher elevations. Columbia spotted frogs are highly aquatic preferring connected pond systems where hydrology and water temperature remains consistent with high emergent vegetation (Welch and MacMahon, 2005), such habitat variables tend to seasonally fluctuate significant in the lower valley. The terrestrial habitat-use by Western toads suggests that highly developed landscapes, like the lower Okanagan Valley, are unsuitable for persistence (Davis, 2002). Few occurrences of Columbia spotted frogs (low availability of wetland systems) and western toads (highly developed terrestrial landscape) in the lower elevation habitats may not be unexpected. Whereas, the Great Basin spadefoot is able to exploit shallow ephemeral wetlands and flooded fields and this may explain the species widespread distribution.

3.4.7 Recommendations and management issues

Natural variations in amphibian population dynamics require long-term studies to investigate suspected declines (Blaustein et al., 2011; Pechmann and Wilbur, 1994). The importance of continuing long-term amphibian studies spanning multi-generations and embracing the attributes and requirements of both aquatic and terrestrial life history stages is paramount in increasing the data reliability and understanding of our changing natural world. Successful conservation strategies need to be at appropriate scales to detect effects of land-use change and incorporate emerging threats. Unfortunately, the appropriate time required for monitoring ecological responses of species and landscapes to change is often much longer than the time scale used in decision making and land-use planning (Magurran et al., 2010). Leaving insufficient data and knowledge gaps to be filled by professional judgment (Martin et al., 2012). Both biodiversity and human behaviour are not static and this is particularly evident in regions with high economic value. Accordingly, dynamic proximate threats require specific conservation policies and programs designed to sustain biodiversity while acknowledging socioeconomic factors or policies favoring agricultural intensification (Mattison and Norris, 2005).

A regional move towards biodiversity planning in the south Okanagan allows agriculture an opportunity to contribute to a sustainable landscape (e.g. South Okanagan Similkameen Conservation Program, Okanagan Wetland Strategy – Okanagan Basin Water Board). And should be supported by the premise that 6% of agricultural area can be withdrawn without negative financial effect to farmer (Lütz and Bastian, 2002). Examining the attitudes and actions of farmers in the south Okanagan Valley, in addition to funding opportunities or constraints, warrant a detailed socioecological and economic study. Generally, the scientific literature identifies a farmers' attitude and actions are most often directly dependent on payment (Genghini et al., 2002; Lütz and Bastian, 2002). Furthermore, farmers who have inadequate knowledge or hostility to agri-environmental measures rarely participate in economic incentives programs despite income benefits (Lütz and Bastian, 2002). Demographic variables, such as absence of wildlife damage to property, presence of a hunter in the family, and youthfulness were positively linked with economic incentives programs (Genghini et al., 2002).

Our study reinforces the need to protect small agricultural ponds and reinforce management practices that increase wetland integrity (i.e. buffers and riparian areas to reduce inputs). Unfortunately, little governmental protection is provided for ponds and wetlands on private land in B.C. and elsewhere. In B.C., if small ponds do not offer adequate fish habitat they are not protected or they are exempt to regulation due to agricultural activities or grandfather clauses (B.C. Riparian Area Regulations, 2006). While this is diminishing with the regularity of aerial Google Map updates, historic mapping methods often missed small wetland features. The rapid seasonal rate of land-use change is particularly effective at diminishing wetland features such as natural landscape contours and associate soil and plant indicators. Further, ephemeral wetlands may have had the greatest risk of infilling on agricultural land due to their potentially long periods of senescence and seemingly uncharacteristic wetland features. Locally, the provincial Conservation Officers / inspectors and the BC Environmental Farm Plan Program and federally the Canadian Food Inspection Agency work with landowners to increase agricultural sustainability while promoting environmental health (http://www.agf.gov.bc.ca/resmgmt/EnviroFarmPlanning/index.htm). Additionally, local NGO stewardship organizations are the primary means of supporting individual landowners towards environmental responsibility and habitat enhancement activities (South Okanagan Similkameen Conservation Program, 2012). The rapid change in land-use practices and a relatively high turnover in land tenure, coupled with a complex mix of stakeholders reinforce the need for continued annual species and habitat monitoring.

Although it was not measured in this study, an emerging threat to high elevation sites in the south Okanagan Valley is related to the cumulative off-road vehicle activities associated with terrestrial degradation (*see* review Arp and Simmons, 2012). Effects after off-road use can result in altered watershed processes, increased wetland drainage, and decreased water quality consequently reduced habitat suitability for aquatic species (Alaska, USA: Arp and Simmons, 2012). An examination among open, closed, or protected off-road treatment groups and amphibian population parameters found no differences, however, the authors attribute the results to seasonal drought conditions in Texas, USA (Hunkapiller et al., 2009). The frequency or extent of off-road vehicles activity and associated impacts as a habitat or species stressor in BC has not been fully assessed. Direct habitat damage due to off-road vehicles at high elevations wetland sites in the south Okanagan containing Great Basin spadefoot, Blotched tiger salamander, or Western painted turtle species has been documented, and in some cases, resulted in charges under the Forest and Range Practices Act (Patton, 2012). Attitudes may be changing since Provincial legislation in 2007 introduced penalties of up to $100,000 and a year in jail for people who willfully damage ecologically sensitive range habitats (Patton, 2012).

There may be an impact of free-roaming cats on amphibians and reptiles, however few studies have been conducted. In the USA, cat predation was reported to kill between 95 and 822 million amphibians annually with documented cases of species extinction (Medina et al., 1999, Henderson et al., 1992 *from* Loss et al., 2013). While data on cats was not recorded specifically in the present study, the detection of cats was often associated with calling disturbance at ponds during auditory surveys that was notably higher in urban areas. As a result, ponds within the lower south Okanagan Valley urban zones may be population sinks posing a threat as species move along a ‘pinched’ urban landscape as part of the north south corridor from Washington State to the greater Okanagan region.

Climate induced wetland habitat loss is an emerging global threat (Ryan et al., 2014) and variable hydrological conditions are anticipated in an arid environment. Flooding dates and hydroperiods are expected to be variable among years and ponds in response to climatic precipitation and temperature patterns. The influence of climate change on hydrology in the South Okanagan predicts increased seasonal temperature and evapotranspiration with resultant increased water stress due to greater growing season demands (Toews, 2007). Increased evapotranspiration in natural wetlands may result in reduced hydroperiod and consequently decreased available time to complete metamorphosis. Agricultural wetland hydrology is more challenging to forecast due to the dramatic fluctuations in response to infiltration and overflow irrigation practices (Toews, 2007) and water management regimes of the Okanagan river channel. While precipitation and topography has been empirically tested to predict ephemeral hydrology (Bauder, 2005; Brooks, 2004), the relative contributions of climate change and land-use practices on hydrological systems are yet to be adequately quantified. Population dynamics for wetland breeding amphibian and reptiles are strongly regulated by hydrology. Changes in wetland hydrology affect population processes (Morey 1998) alter community composition (Schneider 1997) and increase vulnerability to invasion by non-native species (Gerhardt and Collinge, 2003) (*from* Pyke and Marty 2005). Ephemeral wetlands may experience increased hydrological responsiveness that may be tied directly to population dynamics and uncertainty.

While the present work identifies species occurrence, richness, habitat availability and quantifies some threats, the overall understanding of amphibian demographics in the south Okanagan is still relatively lacking. Metapopulation dynamics by species examining the population connections among south Okanagan wetlands is lacking but critical. Conductivity becomes increasingly hard to assess in regions where development is increasing and rapidly changing (Compton et al., 2007) and where species are moving between wetlands as ponds dry or become inundated in response to land-use practices. The existing south Okanagan amphibian monitoring data does not yet allow for the examination of species stability, colonization, or extinction rates at a geographic scale, which is necessary for a true assessment of population trends (Hecnar and M'Closkey, 1997a,c). The low detection of amphibians and turtles in the south Okanagan Valley implicate small and fluctuating populations vulnerable to local extinctions resulting from long-term environmental stress and stochastic demographic factors (Hecnar and M'Closkey 1997a). Some widely mobile species (e.g. Great Basin spadefoot, Pacific chorus frog) may have an imposed metapopulation 'extinction- recolonization' structure due to the ephemeral nature of the valley wetlands but yet be able to persist at a regional scale. Unfortunately species with restrictive habitat availability (e.g. Blotched tiger salamander, Columbia spotted frog) may be less likely to persist at both a local and a regional level.

3.5 CONCLUSION

Particularly at low elevations, the south Okanagan Valley amphibian and turtle populations are relying on the availability of relatively few breeding ponds and a fragmented terrestrial landscape among a mosaic of predominantly agricultural and urban land development. There are many competing demands for land with the need to restore landscapes for the protection of biodiversity and natural capital (Parott and Kyle, 2014). Remaining wetland and natural areas are prone to the continuous fragmentation process and a consequential loss of habitat quality. It is clear for the south Okanagan Valley, that the protection of the remaining wetland mosaic in entirety is vital if there is any hope for long-term persistence of any amphibian or turtle species. The future of amphibian and turtle populations in the south Okanagan Valley relies on monitoring, protection, and active management of all remaining individual wetlands within the mosaic. Further, the spatial arrangement of wetlands, populations dynamics, habitat restoration, including the associated terrestrial habitat needs, provide for extensive unanswered research questions within the valley. Here, we provide amphibian indicator species monitoring data and wetland landscape assessment; the next steps are to incorporate the biological and landscape data to explore strategically located habitat enhancement actions to reconnect priority wetland and terrestrial habitats for amphibian and turtle species.

3.6 ACKNOWLEDGEMENTS

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4.0 RECONNECTING AMPHIBIAN HABITAT THROUGH SMALL POND CONSTRUCTION AND ENHANCEMENT, SOUTH OKANAGAN VALLEY, B.C.

4.1 INTRODUCTION

It has been estimated that, 32.4% of amphibian populations are globally threatened (IUCN categories; Stuart et al., 2004) implying a rate of mass extinction (Wake and Vredenburg, 2008). Habitat loss is considered the most significant factor in this decline; other stressors such as pollution, overexploitation, non-native species, and disease are also contributing (*see* review Blaustien et al., 2011). To counteract these problems, a variety of approaches have been used including translocation and habitat restoration (Calhoun et al., 2014; Pechman et al., 2000; Shulse et al., 2010). In each of these initiatives, the lack of quantification of success or failure of these techniques has often made it difficult to determine if they have made any long-term difference to amphibian populations on a local or broader scale. In Canada, amphibians experience the same stressors as elsewhere along with a shorter growing season and extreme winter temperatures (Lesbarrères et al., 2014). Even with the highest annual average temperatures in Canada (B.C. Statistics http://www.bcstats.gov.bc.ca/StatisticsBySubject/EnvironmentalStatistics.aspx), the south Okanagan Valley, the long-term survival of amphibian populations is uncertain in the lowland valley. All status reports on amphibians at risk in the Okanagan Valley note the impact of habitat loss (Committee on the Status of Endangered Wildlife in Canada, COSEWIC 2001; COSEWIC 2007) These amphibian species are relying on less than 16% of the historic wetlands (Lea, 2008) yet there is recognition that the remaining wetland ecosystem supports unique desert adapted amphibian diversity (South Okanagan Similkameen Conservation Program, 2012).

Within the south Okanagan Valley, British Columbia, low species richness (66% of sites < 2 species) and very low reproductive success in the remaining ponds in this region (67% of sites < 10 individuals of any early life stage) (*see* Chapter 3), suggests that survival of populations may require intervention. While natural ponds are scarce, agricultural ponds in the lower Valley now represent an alternative and perhaps prevalent breeding habitat (*see* Chapter 3), and likely contribute significantly to amphibian population persistence (Knutson et al., 2004). The construction of small ponds that are near to one another and interconnected by relatively natural corridors may be an approach that could support recovery of distinct populations of amphibian species at risk (SAR) occurring in the south Okanagan Valley.

The south Okanagan Valley landscape fits within the theoretical principles of Island Biogeography Theory (MacArthur and Wilson, 1967), 'Single Large Or/And Several Small' (SLOSS/SLASS) protected area management (Diamond, 1976), and the dynamics of Metapopulation Theory (Levins, 1969). However, the value of artificial construction of wetlands to mitigate losses for amphibians has been found to be ambiguously successful (Sutter and Francisco, 1998; Calhoun et al., 2014). Therefore, after constructing and enhancing 21 ponds within the south Okanagan Valley (Fig. 4.1) we tested several hypotheses to determine whether these ponds attracted amphibians, and if they could successfully reproduce in these ponds. We propose the hypothesis to explain the use of constructed and enhanced wetlands as a recovery action to support amphibian SAR:

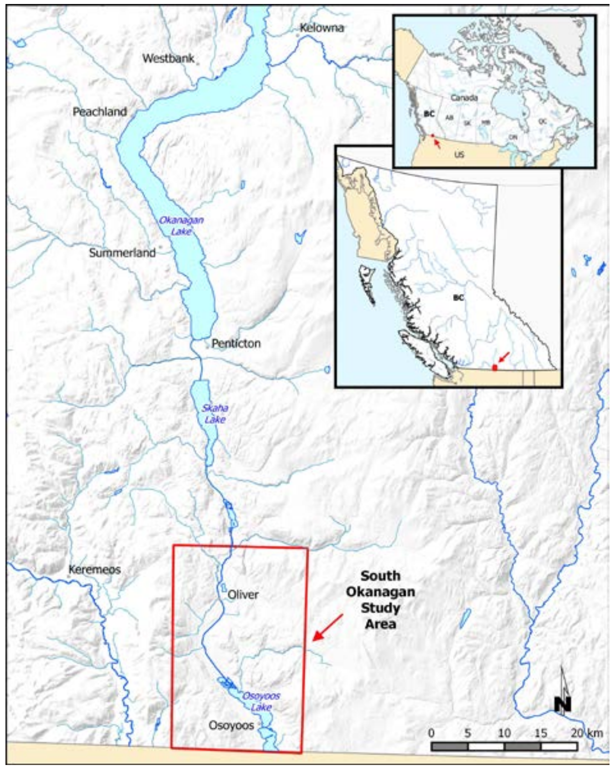

Figure 4.1. Habitat enhancement and small pond construction study area, lowland south Okanagan Valley, British Columbia.

> Ha1 = If amphibians are able to use anthropomorphic breeding sites, then the construction of strategically placed ponds will increase the number of potential breeding sites and habitat area and consequently will be utilized and result in successful reproduction.

We also tested to determine if terrestrial soil habitat characteristics that are required by fossorial species surrounding constructed ponds were similar or different than existing ponds. Our long-term goal is to produce a complex of wetlands where amphibian communities are similar to, or increasing in population size, compared to existing south Okanagan Valley ponds.

4.2 METHODS

4.2.1 Site selection

Sites for wetland pond construction, enhancement, and/or non-native invasive species removal were selected based on known herpetofaunal species data, biological criteria (e.g. dispersal abilities, *see* Semlitsch, 2001; Semlitsch and Brodie, 2003), or historical occurrence records, or in partnership with other non-government organizations (NGO), government organizations (GO) and conservation authorities (i.e. wetland mitigation compensation B.C. Ministry of Transportation and Infrastructure). The partnership arrangements required landowners to sign a 5 to 10 year voluntary stewardship agreement that outlined permissible (e.g. instillation of bat boxes) or restricted activities (e.g. addition of non-native plant or animal species) around the pond. Ideally, the biological and management criteria for choosing sites for wetlands activities are (Fig. 4.2):

- Within 500 m of a known target species breeding population,
- Within 500 m of an additional water way (pond, lake, creek, oxbow),
- A minimum 100 m distance from a roadway.

3.2.2 Wetland habitat creation and restoration

Wetland habitat creation and restorations were classified into one of three classes (Table 4.1):

1. Construction (N = 10): No pre-existing wetland ever existed in the local
2. Enhancement (N = 8): Wetland existed historically and was in filled
3. Invasive predatory species removal (N = 3): Wetland exists with predatory non-native species present

The technique used to artificially construct and enhance wetlands was developed following Biebighauser (2003). All construction and enhancement activities had a site plan to envision layout (e.g. size and shape) and establish conservation management practices (e.g. buffers, spray shed location), site re-vegetation plan, and terms of the stewardship agreement. A preliminary on-site assessment was conducted to ensure appropriate site conditions (i.e. test hole to observe inundation and the depth of fine textured soil, such as clay and silt loam, needed to retain water). Where the substrate was permeable, an alternative design was implemented installing an ‘aquatic safe’ synthetic 45 ml ethylene propylene monomer (EPDM) liner coupled with a geotextile fabric pad to reduce root penetration and absorb wildlife impact (N = 3). In six sites, a registered professional biologist was hired to assess if any SAR might be negatively impacted by the activities. All activities were conducted in the presence of a biologist and when on private lands the owner was present to participate. Construction and enhancement activities were conducted in the fall when the ground was driest and the likelihood of disturbing wildlife was decreased.

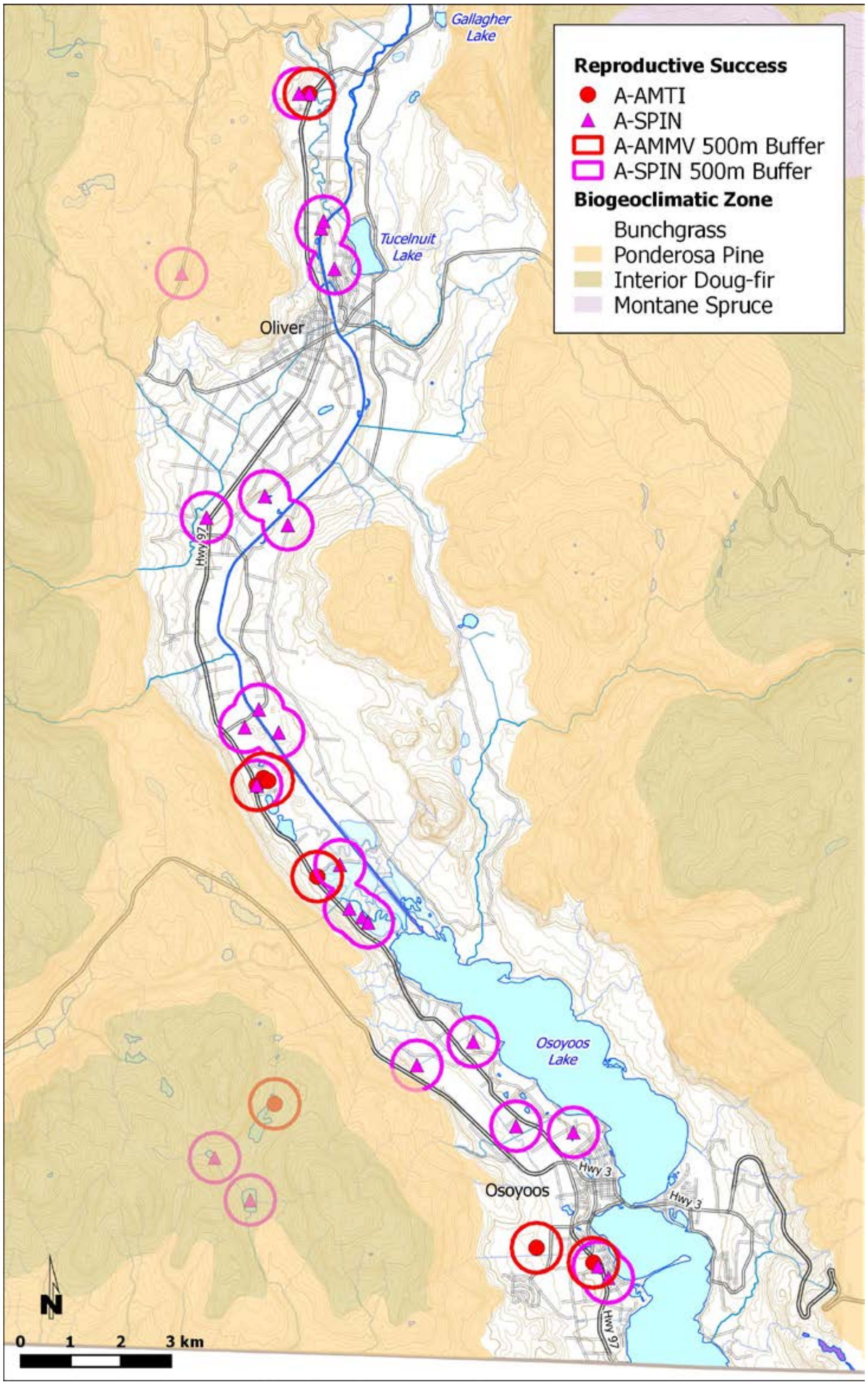

Figure 4.2. Known Great Basin spadefoot and Blotched tiger salamander breeding sites within the south Okanagan Valley, B.C. study area. Breeding success was presumed by the presence of early life stages (e.g. eggs, tadpole/larvae, metamorph). To increase the likelihood of colonization a 500m buffer around breeding sites was used to help prioritize wetland habitat activities. Species codes: AMMV Blotched tiger salamander, SPIN Great Basin spadefoot.

A single exception was made, where coordination with Conservation Authorities (B.C. Ministry, Report all Poachers and Polluters) permitted activities in early spring.

An experienced contractor using either a backhoe or an excavator excavated the sites for the reconstructed wetlands. The general aspects of pond shape and water depth were variable and often constrained by the site geographical, physical properties, and land-use characteristics, when possible the bank was contoured with a 10:1 slope ratio (1 meter of rise for every 10 meters horizontal distance). Design considerations included species-specific priorities. This meant that sites designed to attract Great Basin spadefoots where generally dug less than 0.6m deep relying predominantly on surface run off, whereas ponds designed to attract Blotched tiger salamanders had depths as deep as 1.5 m. The excavated material (surface vegetation and subsurface soil) was reincorporated into the banks (N = 13), or because of restricted land availability or a very high density of non-native invasive plants were either moved to a composter on site (N = 1), or taken off site to the regional dump (N = 4). To reduce erosion and promote soil structure, in addition to reduce weed invasion (Moncada and Sheaffer, 2010), the exposed excavation area was seeded by hand using a winter rye as a cover crop followed by straw mulch (Biebighauser, 2003).

While the construction and enhancement activities to establish ponds were completed in 1 to 3 days effort, the invasive predatory fish (*Carassius auratu,* N = 1 site) were removed using seine nets over two consecutive spring seasons (2008 and 2009) (Ashpole et al., 2011). Removal of American bullfrogs (*Lithobates catesbeiana)* required extensive effort over 8 seasons (N = 2 sites, 2004 to 2011; as described in Lukey 2012).

4.2.3 Wetland monitoring

From late April to mid July, both daytime visual encounter surveys and nighttime auditory surveys were conducted at each site on two to six occasions per season using methods from The B.C. Ministry of Environment, Lands, and Parks (1998). Visual encounter surveys involved searching the wetland shoreline and submergent vegetation zone, in some cases using a dip net. All species observed were counted and the life stage (e.g. egg, larval, metamorphic, hatchling, adult) recorded (Table 4.3). Auditory surveys lasted at least five (uninterrupted) minutes and were recorded using a calling index, where 0 = no calls heard, 1 = calling individuals can be counted, 2 = calls of individuals are distinguishable (some calls overlap), 3 = full chorus, individuals cannot be distinguished (BC Ministry of Environment, Lands, and Parks, 1998). As the detection of any occurrence was valuable, all incidental calls were also recorded. In 2007-2008, and 2014 permanent wetlands (N = 4 sites) adjacent to known Blotched tiger salamander sites were surveyed using baited (canned sardines) minnow traps fitted with flotation noodles. Commercial minnow trap dimensions were 23 x 41 cm with a 3 cm opening on each end and constructed with

0.65 mm galvanized steel mesh and a spring-clip closure. Traps were set at sunset and retrieved early the following morning. All animals were released at the point of capture. Time constrained surveys were conducted by graduate students and the author, whereas effort was often conservatively estimated for trained NGO, GO, and landowner voluntary reports, which represented about 20% of survey effort.

All procedures conducted in this research followed the Canadian Council on Animal Care (Olfert et al., 1993) using approved protocols from Environment Canada (Delta, B.C.) and University of Waterloo (AUP # 1109) Animal Care Committees; research permits were obtained from the BC Ministry of Environment (# PE06-21835; #PE13-87949). To ensure no cross contamination of amphibian disease or transport of non-native species among sites all field equipment was disinfected with 10% bleach solution daily. To further reduce risk, sites with known American bullfrog populations had dedicated site-specific equipment.

4.2.4 Terrestrial soil charachteristics

In 2008, terrestrial substrate suitability to enable fossorial species burrowing - represented by soil compaction - was assessed at a subsample of wetland sites (N_{total} = 21 sites) as a measure of upland terrestrial habitat quality. Using a handheld penetrometer (Cole Parmer #EW-99039-00) the unit was placed upright against the substrate until the calibration mark on the piston was level with the soil; the hole’s depth was then measured as an indicator of compressive strength from 0 to 431 kN/m^2 using a 0 to 4 gradient scale. Wetland sites were classified as conventional orchards e.g. use of conventional farming techniques including inputs, tilling etc. (N = 8), reference e.g. native grasslands or non-grazed fields (N = 6), and constructed (project) ponds (N = 7).

From the pond shoreline, at each cardinal direction, measurements were taken along a transect at 3, 13, 28, and 192 m respectively. At each sampling point, 3 repeated penetrometer measurements were taken and the average recorded. The type of land cover at the sampling points was recorded and included eleven cover classes: water, reed canary grass (*Phalaris arundinacea* L., Poaceae) hay field, new vineyard or recently tilled vineyard, native vegetation, native desert grasses, invasive weeds, couch grass (*Elymus repens* (L.) Gould, Poaceae), cement or gravel, no-till area between the fruit tree lines, and other i.e. garbage, structure.

To compare differences between the three site classifications and with increasing distance from ponds a repeated measures analysis of variance compared site classifications with a multiple comparison post hoc Dunnett test comparing each classification with the reference sites. A second analysis was a univariate test assessing the quadrate class (e.g. hay field), treating distance as independent, followed by a Tukey post hoc analysis to determine which quadrat classes are contributing to differences seen among substrate types. All statistical analyses were performed using Statistica 6.1 (Statsoft, 2003).

4.3 RESULTS

Of the 21 project sites, 13 sites are located within the priority 500 m buffer areas of adjacent breeding sites (Fig. 4.3). A variety of ephemeral and permanent wetlands were designed (Table 1), with 3 sites within proximity and constructed or enhanced specifically for Blotched tiger salamanders (site # 9, 10, 14). Some sites required additional construction, most notably site #11 that was enhanced in fall 2007; it failed to hold water and required an EPDM that liner was installed in fall 2010. Site #21 was the only site that failed outright - it has not successfully held water since construction. Three enhancement sites that filled with water initially are becoming seasonally dry more frequently (site # 9, 10, 15; Table 4.1).

Table 4.1. Summary of wetland site activities and general parameters, south Okanagan Valley B.C., 2006 to 2012. Codes: NGO Non-governmental organization, GO Governmental organization.

Land tenure	Site #	Year	Activity class	Pond dimensions (m)	Shape	Water sub-class	Type of closest water body	Distance to nearest water bodies (m)	Distance to closest known breeding pond (m)	Distance to highway 97 (m)
Private	1	2006	Enhancement	22 x 28	Half moon	Permanent	Creek	118	403	243
NGO	2	2006	Enhancement	20 x 30	Oval	Permanent	Isolated oxbow	35	498	973
NGO	3	2006	Constructed	27 x 32	Oval	Permanent	Isolated oxbow	110	287	828
NGO	4	2006	Constructed	6 x 11	Oval	Ephemeral	Isolated oxbow;	-332	277	185
NGO	5	2006	Constructed	7 x 14	Oval	Ephemeral	Isolated oxbow	-365	116	118
Private	6	2006	Enhancement	14 x 37	Half moon	Ephemeral	Permanent pond	128	145	344
Private	7	2007	Constructed	6 x 8	Oval	Ephemeral	Oxbow	15	385	310
NGO	8	2007	Constructed	8 x 9	Oval	Permanent	Oxbow	35	210	905
NGO	9	2007	Enhancement	17 x 31	Oval	Permanent	Permanent pond	100	55	110
NGO	10	2007	Enhancement	25 x 32	Oval	Permanent	Permanent pond	155	75	110
Private	11	2007	Enhancement; 2010 liner	7 x 55; liner 7 x 21	Teardrop	Ephemeral	Lake	110	1300	125
NGO	12	2007	Constructed - liner	30 x 50	Square	Ephemeral	Ornamental pond	65	65	185
NGO	13	2007	Constructed - liner	4 x 8	Oval	Ephemeral	Ornamental pond	405	405	676
Private	14	2008* 2009*	Non-native species	40 x 100	Oval	Permanent	Permanent pond	210	377	60
Private	15	2009*	Enhancement	17 x 19	Oval	Ephemeral	Ephemeral pond	297	1000	335
Private	16	2009*	Enhancement	50 x 70	Horseshoe	Permanent	Lake	239	664	260
GO	17	2010	Constructed	17 X 42	Teardrop	Permanent	Permanent pond	79	140	82
GO	18	2011	Constructed	40 x 40	Oval	Permanent	Ephemeral pond	240	240	71
Private	19	2004 2011*	Non-native species	62 x 84	Oval	Permanent	Permanent pond	317	360	170
Private	20	2004 2011*	Non-native species	76 x 130	Oval	Permanent	Permanent pond	317	560	402
GO	21	2010	Constructed	32 x 51	Oval	Ephemeral	Permanent pond	75	75	25

*Spring habitat enhancement activities, all remaining activities were conducted in the fall

4.3.1 Wetland monitoring

During sampling, only two of six native amphibian species used the ponds in great numbers (Table 4.2). The Great Basin spadefoot was the most frequent species to colonize (18 of 21 ponds) and successfully produce metamorphic individuals (13 of 21 ponds) (Table 4.3). The Pacific chorus frog colonized a similar number of ponds (16 of 21 ponds) but was less successful at producing metamorphic individuals (7 of 21 ponds). A third amphibian - the Columbia spotted frog - was only observed as producing metamorphic individuals twice at one site (# 1).

Table 4.2. Annual search effort, colonization (presence of any life stage), and metamorphic emergence of amphibian species utilizing constructed, enhanced, and pond sites with predatory non-native species removed (2007 to 2014). Excludes two records of metamorphic Columbia spotted frogs (site # 1) and Western painted turtles (site # 2, 6, 8, 17) presumed to have immigrated from known nearby breeding ponds.

Year		2007	2008	2009	2010	2011	2012	2013	2014
Total # sites		6	14	16	21	21	21	21	21
Auditory search effort (hrs.)		2.25	4.18	4.35	5.13	7.43	5.05	6.11	7.32
Active search effort (hrs.)		4.53	12.33	10.51	10.13	9.24	9.58	8.59	13.44
Great Basin spadefoot	colonized	2	10	11	10	7	13	15	13
	metamorph	1	10	8	5	3	5	5	7
Pacific chorus frog	colonized	3	7	7	7	7	4	6	8
	metamorph	1	0	0	3	2	2	2	3
Total sites colonized		3	11	14	15	12	14	17	14
Total sites with metemorphic		1	10	11	7	4	7	6	8

Table 4.3. Annual wetland colonization by amphibians, south Okanagan Valley B.C. (2007 to 2014). Sites which failed to fill seasonally are indicated as dry. Codes: SPIN Great Basin spadefoot, RALU Columbia spotted frog, PSRE Pacific chorus frog, A-Adult, M-Metamorph, T-Tadpole, E-Egg. Blank cells indicated no species observed. Hatched cells indicate site not yet completed.

Site #	Activity Class	Water Class	Species	2007	2008	2009	2010	2011	2012	2013	2014
1	Enhancement	Permanent	SPIN		A, T, M	A	A	A	A, T, M	A	A
			PSRE		A	A, M	A			A	A
			RALU		M*						M*
2	Enhancement	Permanent	SPIN	A, E, T, M	A, E, T, M	A, E, T, M	A, E, T	A		A	A, T
			PSRE	A, E, T, M	A, E, T	A, E, T	A, E, T, M	A, T, M	A, T	A, T,	A
3	Constructed	Permanent	SPIN		A, E, T, M	A			A, T, M	A, T, M	A, T, M
			PSRE	A	A, T	A, T, M		A, T		A, T, M	A, T, M
4	Constructed	Ephemeral	SPIN			A, T					
			PSRE				A				
5	Constructed	Ephemeral	SPIN				T				
			PSRE		A		A	A, E		A	
6	Enhancement	Ephemeral	SPIN	A, E, T	A, E, T, M	A, E, T, M	A	A, E, T	A, T	A, T	A, E, T, M
			PSRE	A	A			A, E, T			
7	Constructed	Ephemeral	SPIN				A		A	A	A, E, T
			PSRE			A	A, T				
8	Constructed	Permanent	SPIN		A, E, T, M		A, T	A, M		A	E
			PSRE			A, E, T, M	A, T	A, E			
9	Enhancement	Permanent	SPIN		A, T, M	A, E, T	A, E	Dry	A	Dry	Dry
10	Enhancement	Permanent	SPIN		A, T, M	A, E, T, M	A	Dry	Dry	A	Dry
			PSRE		A, T	A, T		E			
11	Enhancement; Liner2010	Ephemeral	SPIN		Dry	Dry	A	A	A	A, E, T	A, E, T
			PSRE		Dry	Dry			A		A
12	Constructed Liner	Ephemeral	SPIN		A, E, T, M	A, E, T, M	A, E, T, M	A, E, T, M	A, T, M	A, T, M	A, E, T, M
13	Constructed Liner	Ephemeral	SPIN		T, M	A, E, T, M	T, M	T, M	T, M	T, M	A, T, M
			PSRE				T	T, M			
14	Non-native Species	Permanent	SPIN		A, T, M	A, E, T, M	A, M	A, T	A	A, M	A, T, M
			PSRE		A	A, E, T	A, T, M	A	A, E, T, M	A	A, T, M
15	Enhancement	Ephemeral	SPIN			A, E, T, M	A, E, T, M	Dry	A, E, T	A - Dry	Dry
16	Enhancement	Permanent	SPIN			A, E, T, M	A	A	A	A	A, T, M
			PSRE			A, E, T, M	A, E, T, M	A, E, T	A, E, T, M	A, E, T, M	A, T, M
17	Constructed	Ephemeral						Dry	Dry	Dry	Dry
18	Constructed	Permanent	SPIN				A, E, T, M	A	A, E, T, M	A, T, M	A, T, M
			PSRE								A
19	Constructed	Permanent	SPIN						A, T	A	A

20	Non-native Species	Permanent	PSRE								A
21	Non-native Species	Permanent	PSRE								
Total number sites				6	14	16	21	21	21	21	21
Number sites colonized				3	11	14	15	12	14	17	14
Number sites with metamorphic frogs				1	10	11	7	4	7	6	8

*Single Columbia spotted frog likely moved through immediately adjacent peat bog from known breeding site (< 17

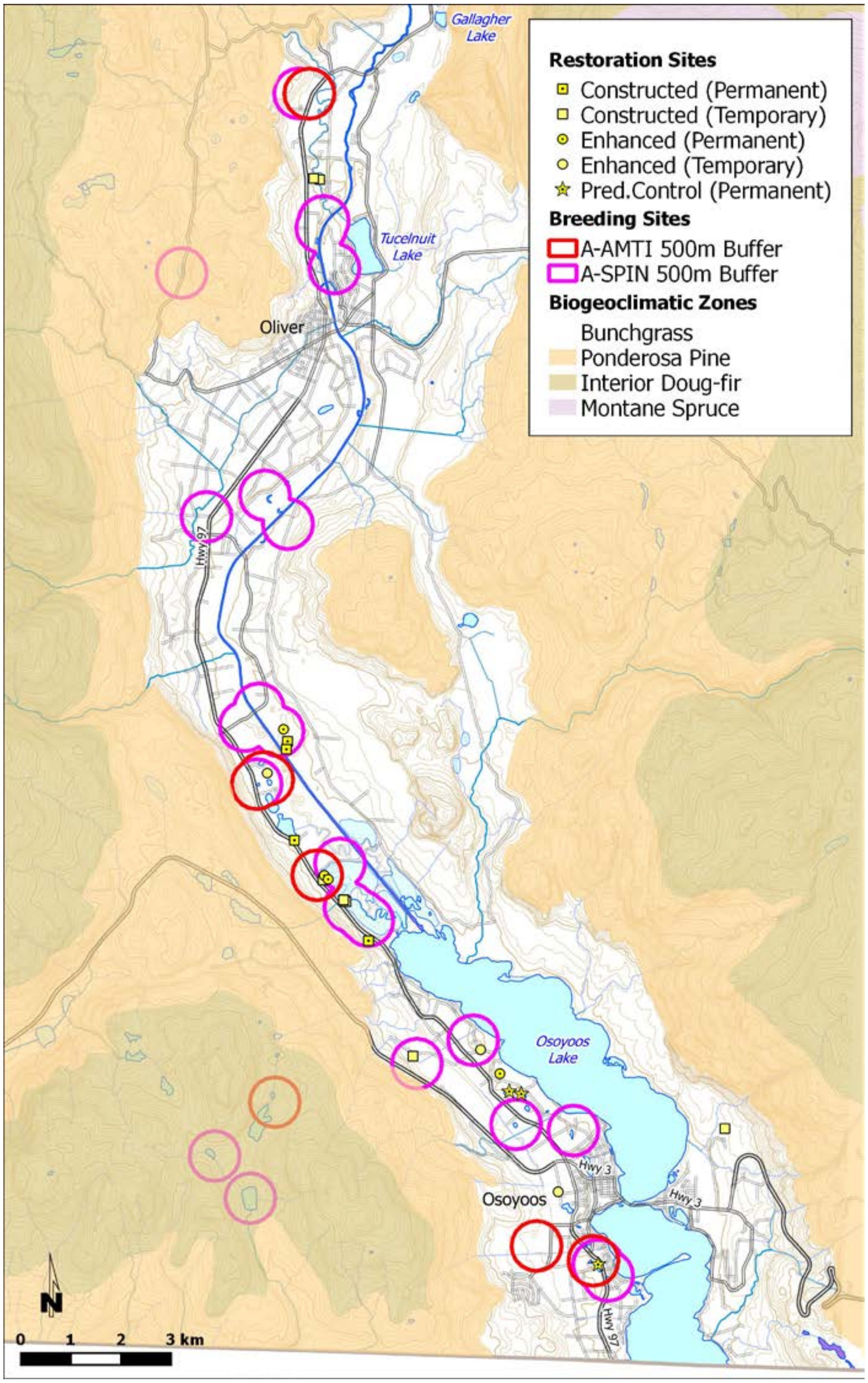

Figure 4.3. Location of wetland habitat activities (construction N = 10), enhancement N = 8, non-native species removal N = 3) in proximity (500 m buffer) to priority Basin spadefoot and Blotched tiger salamander breeding sites south Okanagan Valley, B.C. study area. Species codes: AMMV Blotched tiger salamander, SPIN Great Basin spadefoot.

A maximum of five sites (site # 9, 10, 12, 15, 17) produced Great Basin spadefoot metemorphs in any given year; two sites produced metemophs annually (site # 12, 13). The presence of both Great Basin spadefoot and Pacific chorus frogs metamorphosing in the same year occurred at five sites (site # 2, 3, 13, 14, 16), where as two sites had both species present but only one successfully metamorphosed in any single year (site # 1, 8). Six sites failed to produce metamorphs in any year, but calling adults, eggs, or tadpoles were detected (site # 4, 5, 7, 11, 18, 19). While free of American bullfrogs site #20 did retain non-native invasive fish (e.g. goldfish, *Carassius auratus*; bass, *Micropterus* sp.) and had no recorded amphibian observation. Site #21, failed to hold water (as noted) and also had no recorded amphibian observations.

4.3.2 Soil characteristics

Substrate compaction at reference sites was similar to substrate compaction at our constructed ponds; whereas, substrate at conventional orchard sites (Dunnett: MSE = 2.85, df = 35, $p = 0.018$) was significantly more compact (Tukey: $F(2, 35) = 7.12$, $p = 0.003$) (Figure 4). Further, substrate compaction increased significantly with distance (Tukey: $F(6, 105) = 2.24$, $p = 0.05$) from agricultural ponds (Fig. 4.5). Among quadrate classes, couch grass (MSE 1.83) and cement/gravel (MSE 4.76) substrates were significantly more impenetrable (Tukey: $F(7, 257) = 136.19$, $p < 0.000$), compared to all other quadrate classes which were similar to each other (Dunnett: MSE 0.27 to 0.66).

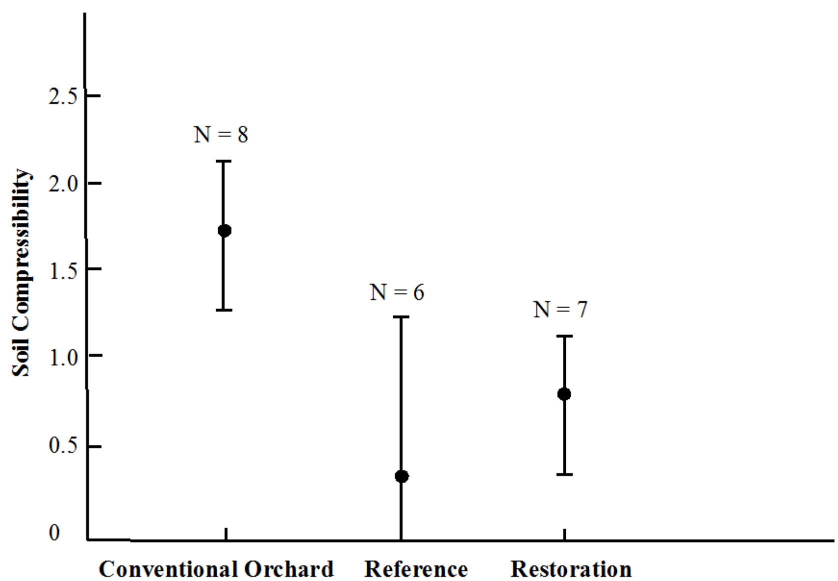

Figure 4.4. Substrate compaction at reference sites was similar to substrate compaction at our constructed ponds; whereas, substrate at conventional orchard sites was significantly more compact. Bars denote 95% confidence intervals.

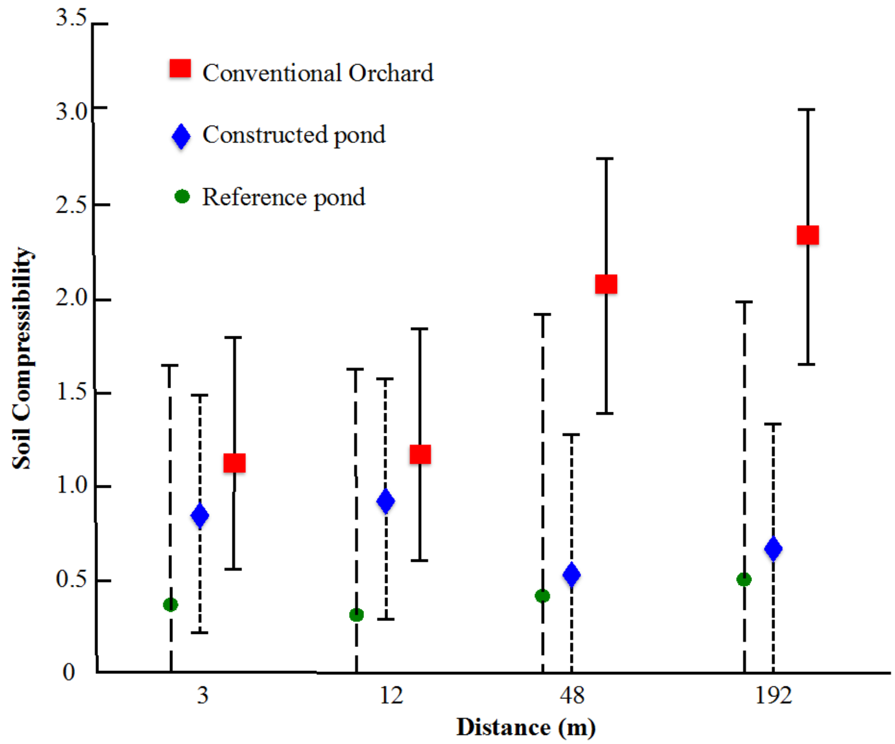

Figure 4.5. Soil compaction varied significantly with distance among a subsample of constructed, conventional orchards and reference ponds with compaction increasing with distance among conventional orchards and remaining consistent among reference sites (Tukey: $F(6, 105) = 2.24$, $p = 0.05$) from ponds. Bars denote 95% confidence intervals.

4.4 DISCUSSION

Our 'small pond' project supports our hypothesis that strategically constructed and enhanced ponds can provide breeding habitat aiding in the Species at Risk (SAR) recovery of Great Basin spadefoots, in the lower south Okanagan Valley. We conducted wetland monitoring, ranging between two to eight years post construction and enhancement activities. We have evidence of Great Basin spadefoot colonization at 86% of ponds (18 of 21 ponds), with confirmed metamorphosis at 62% of ponds (13 of 21 ponds). This was after we conducted wetland monitoring, ranging between two to eight years post construction and enhancement activities; we highlight this because too many projects have no monitoring and we have several with eight years post-construction. We will, continue monitoring as we recognize that project outcomes can change through time. Nonetheless, we contend that our results now support a scaled-up implimentation of restoration management that can be applied throughout the Okanagan Valley and a broader reccommendation that wetland construction and enhancement can mitigate anthropogenic stressors in degraded landscapes for pond species.

While the detection of adult herpetofauna indicated successful dispersal to many ponds and the presence of early life stages suggested successful mating, our measure of pond success was the production of metamorphs as an indicator of habitat quality that can support sustaining populations. Survival was not measured, but one study estimated survival from egg to metamorph in anurans to be 0.7 to 1.3% (Sahara frog species, Bellakhal et al., 2014). Studies examining metamophs at constructed ponds compared to references sites have shown variable outcomes beyond simple presence, including altered community structure, and variability in metamorphic body size depending on species (Morey, 1998; Pechmann et al., 2000). Similarly, some authors contribute differences in metamorphic development to be due to hydrologic regimes, pond size, substrate, vegetation, and surrounding terrestrial habitats and the limited availability of species to colonize (Pechmann et al., 2000; Knutson et al., 2014). While not measured here, variation in similar habitat characteristics have been anicdotally observed and the limited availability of colonization is hypothesised for the Blotched tiger salamander (Ashpole et al., 2011).

The availability of suitable terrestrial, aquatic, and connecting habitat corridors is fundamental to amphibian reproductive success and long-term population persistence (Roe and Georges, 2007; Semlitsch, 2000). Restoration efforts may be marginalized if only a single habitat is considered, which is a common occurrence when working with rare and specialized species. Great Basin spadefoot ecological research has primarily focused on associations with the aquatic breeding habitat (Nyström, 2002; Greenberg and Tanner, 2004; Baughman and Todd, 2007). With Great Basin spadefoots preferring deep, large ephemeral wetlands, with high temperatures, and increased total phosphorus and oxygen levels (Nyström, 2002). Great Basin spadefoots tend to prefer wetlands with low-density emergent vegetation, reduced canopy, and the presence of accessible pathways to the water’s edge (Greenberg and Tanner, 2004; Baughman and Todd, 2007). This implied preference for open wetlands, with a tendency to be warmer compared to cattail wetlands or wetlands with dense riparian vegetation and appears consisted with the habitat preferences observed for Great Basin spadefoots in the south Okanagan Valley.

A high priority when building wetlands is the ability to hold water and achieving the length of pool duration required for larval development (Morey, 1998); this may seem trite but improper design for such basic outcomes can doom projects - even we had one pond fail completely. Amphibians breeding in our nine ephemeral wetlands will likely experience increased potential risk of larval mortality due to seasonal drought. Sites relying strictly on surface runoff were frequently observed dry and increasing hydroperiod in a natural environment is challenging, particularly when restoring ephemeral wetlands. On the other hand, dry ponds after metamorphosis has some benefit in the south Okanagan Valley and elsewhere in that it discourages successful colonization by invasive species such as American bullfrog, whose early life stage development is dependent on readily available aquatic environments for multiple seasons.

The life history of amphibians that inhabit an array of Okanagan wetland types needs to address both the aquatic larval stage and the terrestrial adult stage of species. While it is suggested that surrounding habitat features might not influence pond species diversity (Knutson et al., 2004), terrestrial habitat and surrouunding pond clusters have been landscape indicators in some species occuence (Scheffers and Paszkowski, 2013). The impact of habitat modification and the availability of loose sandy soil, required by burrowing amphibian species, was significantly greater and often impenetrable around existing agricultural wetland sites and sites adjacent to road ways or developed areas compared to our reference and project site ponds. Additionally, the terrestrial area around wetlands represent core habitats and a biologically meaningful (i.e. daily movements, refuges) buffer ranges from 159 to 290 m (based on movement ranges of 32 amphibian species, Semlitsch and Brodie, 2003). Within Semlitsch and Brodie's (2003) recommended buffer range, amphibians in more than half of our wetlands encounter a two-lane highway, which may influence species behaviour (Mazerolle, 2003) and increase probability of mortality (Hels and Buchwald, 2001) and reduce colonization.

Relatively little is known about the ecology, trophic interactions, foraging habitat, or over wintering habitat of the Great Basin spadefoots and even less about Blotched tiger salamanders. Topics of dispersal, home range, and corridor-use are only recently being explored in the northern part of Great Basin spadefoot range, where mean migratory movements are significantly longer than previously thought with mean movements of 690 m (range of 75 to 2350 m; Richardson and Oaten, 2013). If Great Basin spadefoot movement ranges are simlar for our study area, then all ponds within the lower south Okanagan Valley are bisected by highways and fragmented by multiple land uses. Similarly, all lowland ponds are within one kilometer of wetlands where a negative association has been found between the presence of tiger salamanders (*Ambystoma. t. tigrinum*) and linieal distance of paved roads (Porej et al., 2004).

The critical terrestrial habitat element in the south Okanagan Valley for burrowing amphibians is the sandy soil characteristics of an arid desert ecosystem. The soil compaction around conventional orchard wetlands was often greater than 287.3 kN/M^2 (3 tons per sq ft.), which likely impedes fossorial species movment through agricultural lands and urban developed. Intense agricultural practices (high inputs, monocrop culture, soil compaction) and urbanization are the mechanisms driving the loss and transformation of critical native desert habitat (Lea, 2008) at an ongoing rate that has not been quantified.

Loss and transformation of native soils in the south Okanagan (Iverson, et al., 2008) represents a serious issue for the maintenance of ecosystem function, particularly for burrowing species (Gray, et al., 2004; Carisio, et al., 2014). Sandy soils are a common habitat element to the upland terrestrial habitat of the Blotched tiger salamander (COSEWIC, 2001) and the Great Basin spadefoot (Jansen et al., 2001; COSEWIC 2007; Richardson and Oaten, 2013). Uniquely adapted to the arid desert environment, these amphibian species are long-lived, nocturnal, burrowers who spend greater than 85% of their life limited to terrestrial habitat with loose sandy soils (Janson et al., 2001). Similarly the common habitat element of the Gopher snake (*Pituophis catenifer deserticola,* Colubridae, Squamata, Blainville, 1835), Burrowing owl (*Athene cunicularia,* Strigidae, Strigiformes, Molina, 1972), American badger (*Taxidea taxus jeffersonii,* Mustelidae, Carnivora, Schreber, 1777), and the Great Basin pocket mouse (*Perognathus parvus,* Heteromyidae, Rodentia, Peale, 1848) is sandy soil, further all are federally listed as endangered or threatened.

Protecting the unique biodiversity of the south Okanagan will require wide scale protection of soil habitat element within a landscape management framework that incorporates all systems and spectums of transformation (Hobbs et al., 2013) . While our intention was to select project sites within areas of loose sandy soil, only 12 of 21 sites met this criteria, with remaining sites within agricultural lands where farming practices likely contribute to soil compaction. As the sandy soil landscape is altered and compacted via urban and agricultural use, there are fewer and more sparsely distributed pockets of suitable burrowing microhabitats. In urban environments, well-drained sandy soils are being replaced with solid asphalt, grass sod, ornamental plants, and other materials unable to provide suitable refugia for burrowing species (Brooke and Todd, 2007).

Laboratory and field trials have established that adults cannot successfully burrow into sod, and less so in gravel (Jansen et al., 2001). Additionally, energy costs and predation exposure with increased or failed burrowing increase significantly under unsuitable substrate conditions compared with loose soils. Newly metamorphosed spadefoots (*Scaphiopus h. holbrookii*) were more sensitive to substrate modifications: In addition to sod and gravel, metamorphosed spadefoots could not burrow into water-saturated soils (Jansen et al., 2001). Similar experiments looking at juvenile substrate preferences found a higher degree of specificity in substrate selection required for successful burrowing (Baughmann and Todd, 2007) compared to adult spadefoots (*Scaphiopus h. holbrookii*). Changes in yearly juvenile spadefoot survival rate have the largest effect on population persistence (Graham, 2002). Reinforcing the importance that the restoration of landscapes incorporates substrates accessible by juvenile spadefoots.

Comparatively, agricultural microenvironments that often favour crops requiring well-drained soil, similarly undergo significant and constant disturbance through tilling, soil compaction, and impervious ground cover of non-native grasses or weeds (Baughmann and Todd, 2007; Nyström et al., 2007). In a study examining recorded calling spadefoots at breeding sites, where upland habitat was recently developed or had increased agricultural activity, the likelihood of local population disappearance increased significantly compared to reference sites (Nyström et al., 2007).

As substrates in urbanized and agricultural areas become increasingly impervious the likelihood of spadefoots being excluded from these areas increases. The existence or provision of sandy substrates with native vegetation enhancement is an absolute necessity when restoring spadefoot habitats and corridors (B.C. Amphibian and Reptile Guidelines, 2014). However, restoration may be augmented if spadefoots are similar to other highly terrestrial amphibian species: they may make use of artificial or constructed burrows by other species. These opportunistically used burrows may allow for movement across a greater range of substrates. As such, artificially constructed and strategically placed burrowing sites should be explored as a feature for restoration sites with modified substrates.

4.5 CONCLUSION AND RESEARCH NEEDS

The enhancement and restoration of wetlands involves a technical challenge within a dynamic social process. One of the technical challenges experienced was the augmentation of constructed wetlands with water, which may appear a straightforward solution and in some cases feasible, however augmenting water is neither cost effective nor a self-perpetuating system. The project ponds located within agricultural lands benefited from an increase in surface water from irrigation and the availability of water infrastructure to artificially augment the pond. The timing of breeding choruses and subsequent egg laying in agricultural ephemeral ponds were connected to the timing of irrigation and have similarly been observed in other regional ponds (Ashpole et al., 2014). Two of our liner ponds located on the upper benches in natural habitat have accessible water sources and are regularly augmented with water to ensure metamorphic success. However, in these two cases the landowners wait till nearby choruses begin and then augment the ponds and do so until development is completed, representing a more natural breeding chronology. While the initial intention was for ephemeral wetlands to perform as natural systems, it is realized that the manipulation of water has become a recommended approach to climate change mitigation (Shoo et al., 2011). Further, our constructing permanent wetlands adjacent to known tiger salamander sites did not result in any observation of colonization, which might be a lag in dispersement as a result of high site fidelity to natal ponds (e.g. Eastern tiger salamander *Ambystoma tigirium*, Ngo et al., 2009) with an estimate of 20% dispersal to new ponds (e.g. California tiger salamander *Ambystoma californiense*, Trenham 1998).

Great Basin spadefoots are dependent on both aquatic and terrestrial habitats for long-term survival. Fossorial amphibians are encountering landscapes with increasingly modified habitats on all fronts: including the lack of breeding ponds and increasing threat of impervious soils. Restoration efforts require an approach that addresses the unique aspects of species ecology. But, there is a need to go beyond a single species strategy with a narrow focus to address the needs of wetland mosaics and complex ecosystems (Lindenmayer et al., 2008). The ongoing local support and recognition for small wetland conservation continues (Ducks Unlimited Canada, 2013; SOSCP, 2012), but requires protection of identified natural aquatic and terrestrial habitat elements of the south Okanagan Valley under policy. A recent evaluation of the Okanagan wetland ecosystem provides an estimated $314 million/year in regional natural capital and ecosystem services (Parrott and Kyle, 2014) which should place the value of restoration and conservation high. Both the social and ecological benefits of restoration can be significantly enhanced when spatial analysis of stressors and ecosystem services are combined (Allan et al., 2013) and such an approach is currently being implimented in the Okanagan valley (e.g. Ecological Goods and Services Assessment, Okanagan Basin Waterboard).

Five strategic research needs are required to increase the structure and function of the constructed and enhanced ponds compared to existing wetlands and to work towards identifying and developing methodologies to over come restoration thresholds (Hobbs, 2007) in highly degraded ecosystems. First, assessing metamorphic success in regional wetlands is important to determine if our project ponds are responding comparably or whether they have limited suitability (e.g. generalist species, Ferreira and Beja, 2013). Second, amphibian species emerging from both project ponds and regional wetlands need to be assessed for measures of fitness, particularly in relation to wetland and landscape structure to ensure local and restored wetlands contribute to a healthy population. Third, greater research in understanding the movement and habitat needs of the Blotched tiger salamander is required to effectively restore suitable breeding habitat. Fourth, project sites require greater microhabitat heterogeneity, namely the supplementation of natural refuge sites to reduce thermal (Shoo et al., 2011) and predatory stress while support loose soils (Carisio et al., 2014). Lastly, the physical wetland characteristics (Knutson et al., 2004) and/or landscape parameters (Pechmann et al., 2000; Lindenmayer et al., 2008) associated with multi-species reproductive success are need as a guide for future projects that benefit a greater breadth of species.

The dynamic social processes involved with private land stewardship with NGO and GO partnerships are influenced by complex motives, values, and attitudes towards the environment. Within and among stakeholder groups there was a unique set of communication challenges, in both scope and scale of understanding. In most cases, our collaborative engagement with local communities help fostered a positive comitment to environmental repair, with hopes of enabling a resilient social ecological environments (Goldstein and Butler 2010). The use of local communities in environmental decision making can be critical for success of long term restoration goals (Lynam et al., 2007, Raymond et al., 2008). Stakeholder participation can facilitate higher decision making in regards to project implementation and design, and various participatory tools (e.g. Community Value Mapping) have been created to determine when and how this local knowledge is needed (Reed 2008). While many aspects of the operative management and ecological success seems to have been achieved, the longterm socio-ecological success of the restoration project will require continued exploration.

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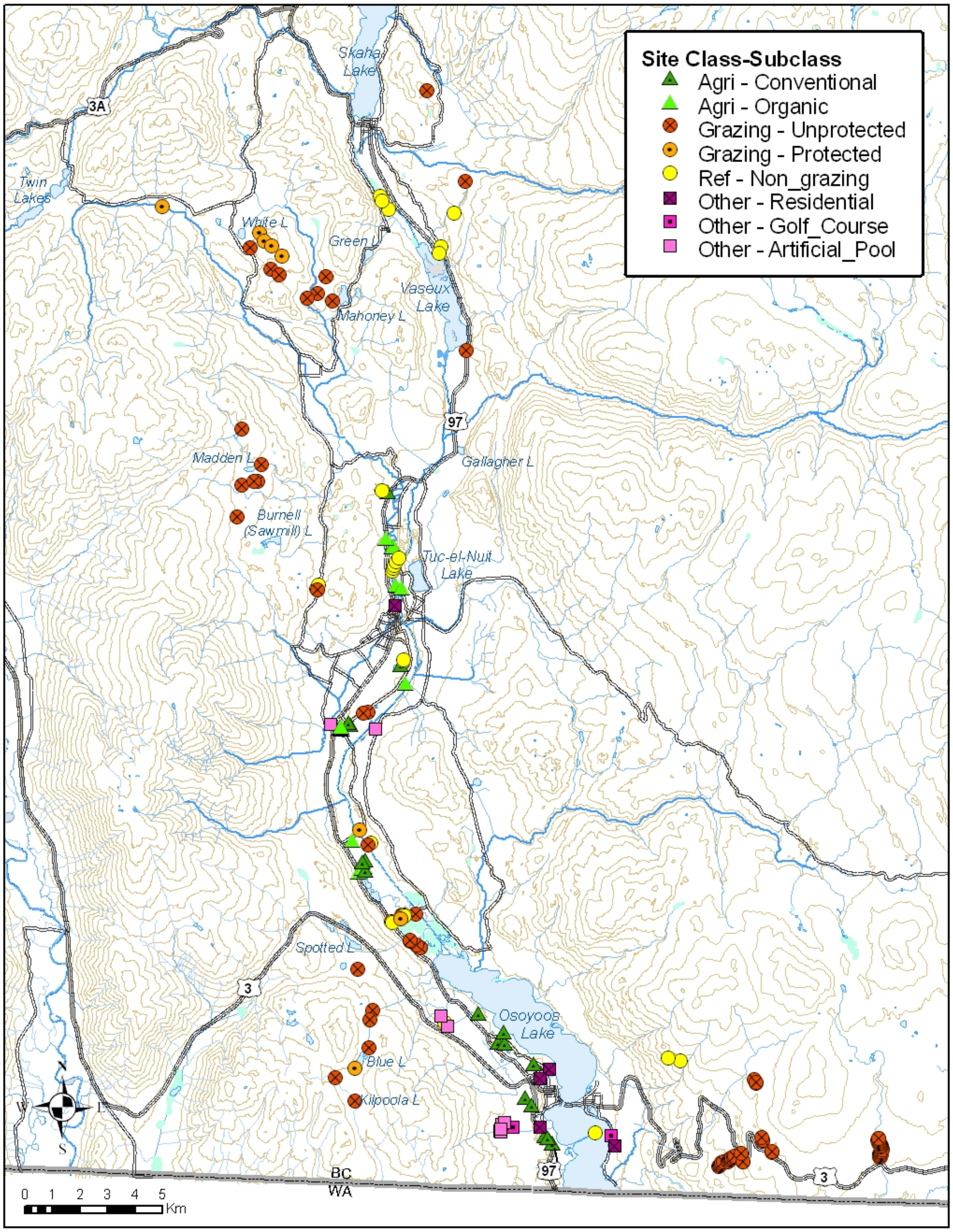
APPENDICES

3.1 Discrete wetland sites were classified based on their dominant land-use and then sub-classified according to land-use practices (sub-class), including reference/ non-grazed (N = 10 sites); agricultural (conventional orchard N = 18 sites; certified organic orchard N = 14), livestock grazed (pond protected from livestock with fencing N = 10 sites; pond unprotected and livestock water access N = 39 sites), and miscellaneous sites (residential N = 7; artificial pools or ponds N = 4; ponds in golf courses N = 2) south Okanagan valley, B.C., 2003 to 2006. Road records (highway N = 48, primary road N = 46, secondary road N = 36) and detections along the river channel (N = 164 records) not indicated.



3.2. With-in site mean (standard deviation) habitat parameters at a subsample of wetland sites ($N_{2003\text{-}2004}$ = 39 sites) by land-use classification, elevation and water permanency, south Okanagan Valley, B.C. (A). Categorical frequency counts by land-use subclass of anthropogenic impacts detected at wetland sites ($N_{2003\text{-}2006}$ = 108 sites) south Okanagan Valley, B.C. For example, 18.5% (or 20 of 108 sites) of sites had water withdrawal or discharge present with the greatest occurrence located in 9 of 16 conventional sites (B). Frequency counts of anthropogenic impacts, including withdrawal or discharge, infilling, garbage, introduced invasive species, agricultural input (e.g. pesticides, herbicides), nutrient input (e.g. unrestricted livestock, turf fertilization), and artificial construction detected at wetland sites ($N_{2003\text{-}2006}$ = 108 sites) south Okanagan Valley, B.C. For example, 12% (or 13 of 108 sites) of sites had zero impacts detected with protected grazing sites having the least likely occurrence of any impact (C).

A)

Site Classification	N_{sites}	Perimeter (m)	Center depth (m)	Distance to closest road (m)	Distance to crop (m)	Riparian-edge vegetation (m)	Emergent vegetation (m)	Sub-emergent vegetation (m)
Land-use subclass								
All	39	292.8 (209.4)	2.00 (1.42)	212.6 (253.7)	10.86 (8.81)	6.23 (7.3)	8.58 (13.41)	4.12 (8.42)
Conventional	14	283.3 (174.70)	2.48 (1.5)	53.4 (34.7)	9.23 (8.71)	8.2 (8.68)	7.25 (5.37)	3.54 (4.5)
Organic	13	203 (105.2)	1.88 (1.37)	224 (165.2)	14.18 (8.22)	5.5 (4.23)	10.37 (21.04)	0.76 (1.65)
Unprotected grazing	7	447.9 (235.4)	0.81 (0.67)	360.3 (304.1)	na	0	0.77 (1.65)	13 (16.63)
Protected grazing	3*	313.9	2.72	560	na	13.7	6.52	2.37
Golf course	1**	91	2.8	244	na	0	0.2	0
Residential	1**	862	2.75	15	na	13.4	0.25	4
Elevation and water permanency								
High elevation temporary	6	460.8 (269.8)	1.22 (0.68)	420. 2 (284.4)	na	1.1 (2.78)	17.61 (19.46)	15.17 (17.1)
High elevation permanent	2*	395	3.78	844.5	na	8.6	9.6	1.8
Low elevation temporary	9	200.2 (109.7)	0.66 (0.78)	121.4 (188.2)	na	8.41 (12.09)	10.13 (22.62)	2.97 (5.55)
Low elevation permanent	22	268.5 (207.9)	2.61 (1.25)	135.9 (147.2)	na	6.51 (5.07)	5.34 (5.01)	1.81 (2.22)

*averages and ** absolute numbers taken due to small sample sizes

B)

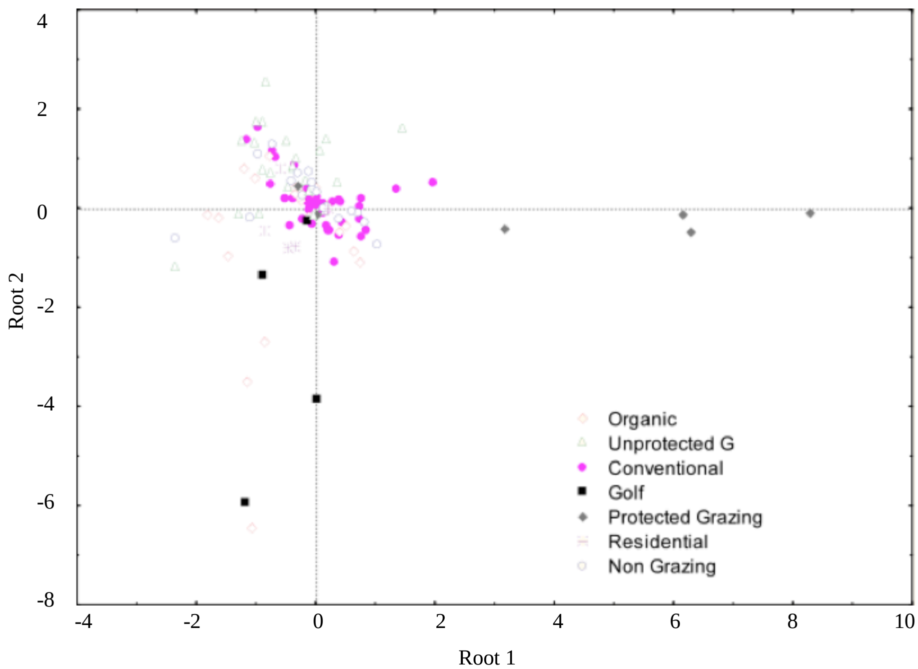
Site Classification	N_{sites}	With-drawal / discharge	Infill	Garbage	Invasive species	Agri-input	Nutrient input	Artificial construction
Land-use subclass								
All	108	20 (18.5%)	20 (18.5%)	20 (18.5%)	21 (19.4%)	16 (14.8%)	64 (59.2%)	40 (37.0%)
Conventional	16	9	9	11	7	16	16	4
Organic	14	4	5	1	1	0	14	7
Unprotected grazing	31	3	1	4	5	0	31	7
Protected grazing	9	0	0	0	0	0	0	2
Non-grazing	23	1	3	1	4	0	0	12
Golf course	3	1	0	0	1	0	3	3
Residential	7	2	2	3	3	0	0	1
Artificial (cement liner)	5	0	0	0	0	0	0	5

C)

Site Classification	N_{sites}	Number of anthropogenic impacts by frequency count and percent (%)						
		0	1	2	3	4	5	6
Land-use subclass								
All	108	13 (12.0%)	26 (24.1%)	16 (14.8%)	5 (4.6%)	9 (8.3%)	6 (5.6%)	2 (1.9%)
Conventional	16	0	0	0	2	6	6	2
Organic	14	0	2	8	2	2	0	0
Unprotected grazing	31	0	21	7	1	2	0	0
Protected grazing	9	7	2	0	0	0	0	0
Non-grazing	23	5	16	3	0	0	0	0
Golf course	3	0	0	2	0	1	0	0
Residential	7	1	1	3	1	0	0	0
Artificial (cement liner)	5	0	5	0	0	0	0	0

3.3. DFA root structure for main land-use classes for water chemistry samples, south Okanagan Valley, B.C., 2003 to 2006 (A). Root 1 was associated with higher pH, which was highest at the protected grazing sites relative to all the other sub-classes. Root 2 was negatively associated with nitrate, nitrite, and total nitrogen, which were highest at the golf course compared to other groups (B). DFA Root 2 structure for nitrite and total nitrogen concentrations (mg/L) for main land-use classes for water chemistry samples (Wilk's $\lambda = 0.218$, $F_{(90, 743.31)} = 2.57$, $p = 0.000$)(C).

A)

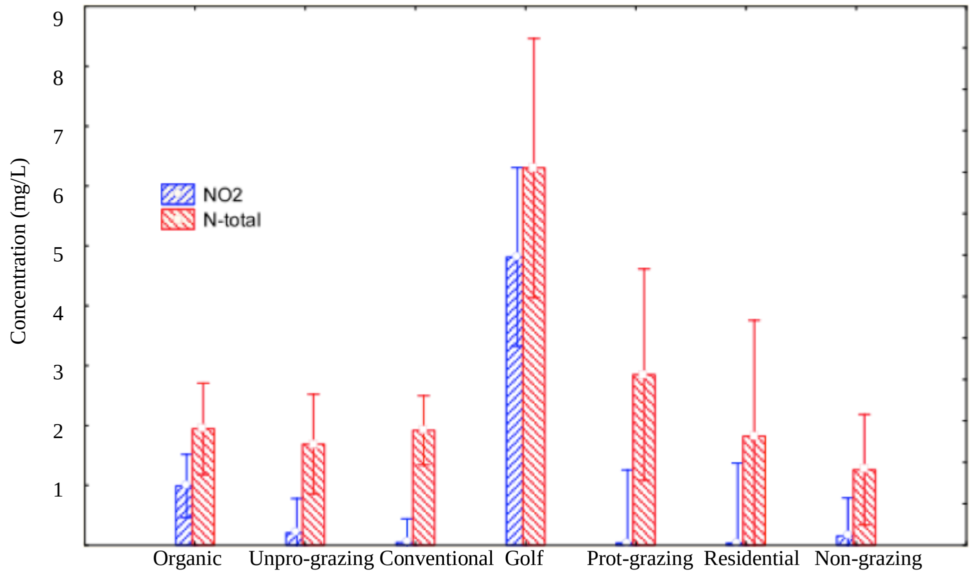


B)

Water Chemistry Parameter	Root 1	Root 2
BOD	-0.10	0.20
Cl	-0.09	-0.13
F	-0.01	0.09
SO_4	0.26	0.06
Br	0.14	0.10
NO_2	-0.15	**-0.73***
NO_3	-0.08	**-0.57***
PO_4	-0.06	0.20
pH	**0.40***	-0.04
Conductivity	0.25	0.02
Turbidity	-0.04	0.08
NH_3	-0.05	0.07
N-total	0.04	**-0.47***
o-PO_4-diss	-0.09	0.16
P-total	-0.09	0.15

Bolded (*) values identify roots with highest correlations.

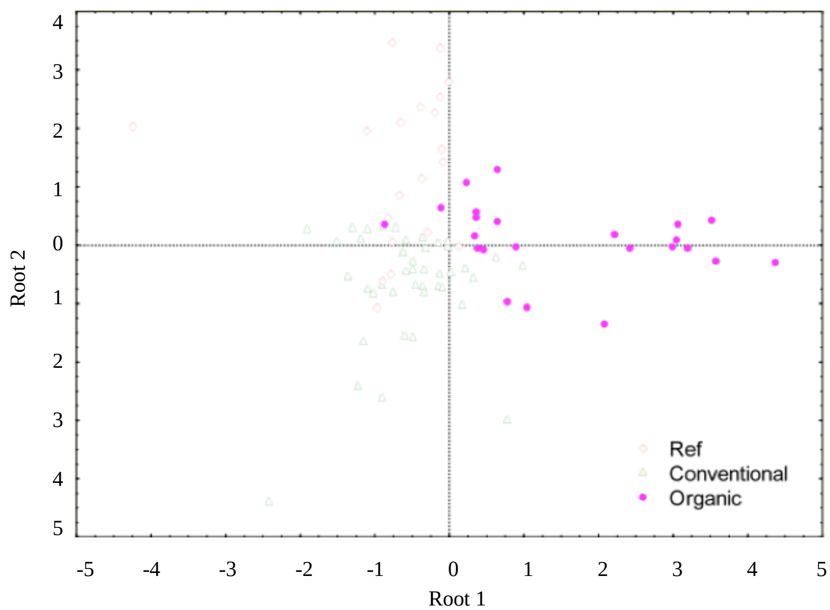
C)



Concentration (mg/L)
9
8
7
6
5
4
3
2
1
NO2
N-total
Organic
Unpro-grazing
Conventional
Golf
Prot-grazing
Residential
Non-grazing

3.4 DFA root structure for agricultural and reference sub-classes, south Okanagan Valley, B.C., 2003 to 2006 (A). Agricultural and reference sub-class land-use DFA factor structure matrix correlation variables and canonical roots. Root 1 was associated with higher sulfate and nitrite at the organic sites relative to conventional and reference sites whereas, Root 2 was negatively associated with turbidity (B).

A)



B)

Water Chemistry Parameter	Root 1	Root 2
BOD	-0.133	0.080
Cl	0.097	0.187
F	-0.183	0.221
SO_4	**0.362***	-0.063
Br	0.096	-0.219
NO_2	**0.345***	0.036
NO_3	0.086	-0.109
PO_4	0.026	-0.187
pH	-0.120	-0.049
Conductivity	0.323	-0.070
Turbidity	-0.128	**-0.299***
NH_3	-0.042	-0.206
N-total	0.144	-0.094
o-PO_4-diss	0.205	-0.133
P-total	0.076	-0.202

Bolded (*) values identify factors with highest positive loadings.

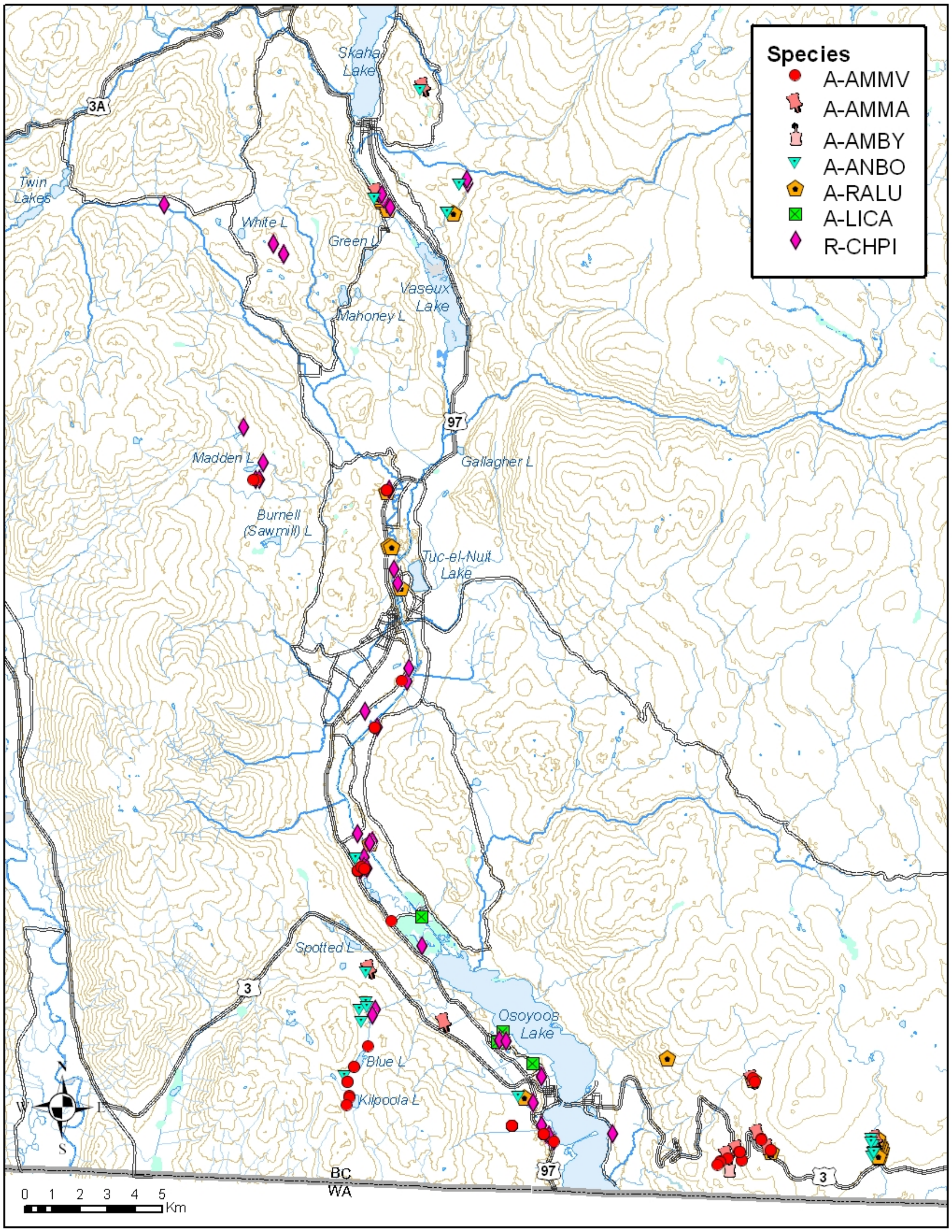
3.5. Factor analysis and maximum likelihood factor loadings explained 72.4% of the total proportional variance among water quality variables from agricultural and reference land-use sub-class sites, south Okanagan Valley, B.C., 2003 to 2006.

Water Chemistry Parameter	**Factor 1**	**Factor 2**	**Factor 3**	**Factor 4**
BOD	0.14	0.03	0.93*	0.04
Cl	0.38	0.01	0.19	0.15
F	-0.20	-0.07	-0.04	-0.11
SO_4	**0.94***	-0.05	0.14	-0.06
Br	0.50	0.01	**0.80***	0.10
NO_2	0.10	-0.03	0.09	**0.98***
NO_3	-0.02	-0.02	-0.02	**0.54***
PO_4	-0.04	**0.92***	-0.01	-0.02
pH	0.22	-0.12	0.13	-0.13
Conductivity	**0.90***	-0.05	0.41	0.04
Turbidity	0.25	0.02	0.94*	0.09
NH_3	0.32	0.02	0.82*	0.10
N-total	0.22	0.02	0.47*	**0.69***
o-PO_4-diss	-0.05	**0.99***	-0.02	-0.03
P-Diss	0.05	**0.95***	0.28	0.01
P-total	0.16	**0.53***	0.82*	0.07
Explained variance	2.44	3.05	4.28	1.83
Proportional total	0.15	0.19	0.27	0.11

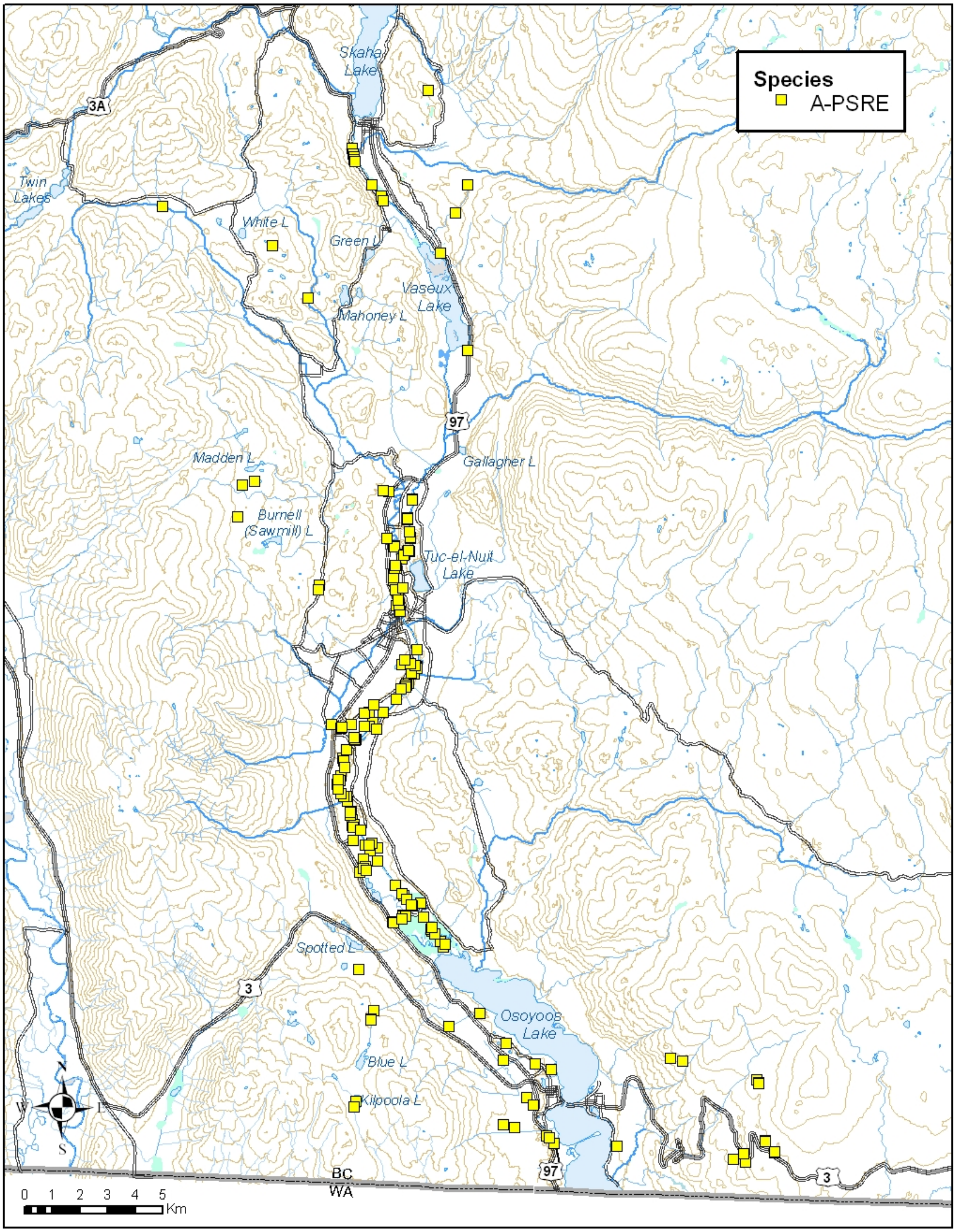
Bolded (*) values identify factors with highest positive loadings.

3.6 Occurrence data for Blotched tiger salamander (A-AMMV), long-toed salamander (A-AMMA), unknown ambystoma salamander species (A-AMBY), western toad (A-ANBO), Columbia spotted frog (A-RALU), American bullfrog (A-LICA), Western painted turtle (R-CHIP) (A), Pacific chorus frog (A-PSRE) (B), and Great Basin spadefoot (A-SPIN) occurrence data, south Okanagan Valley, B.C., 2003 to 2006 (C).

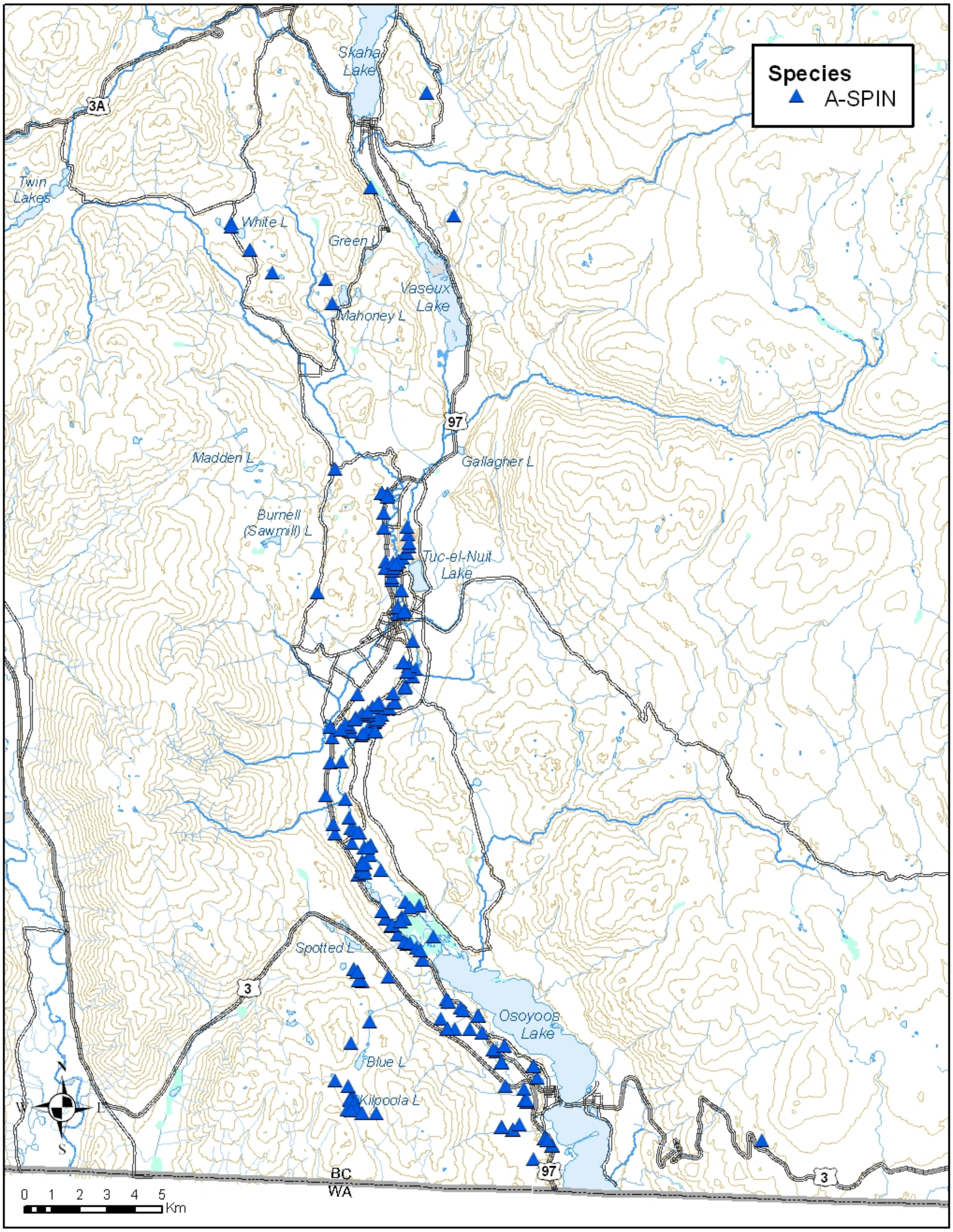
A)



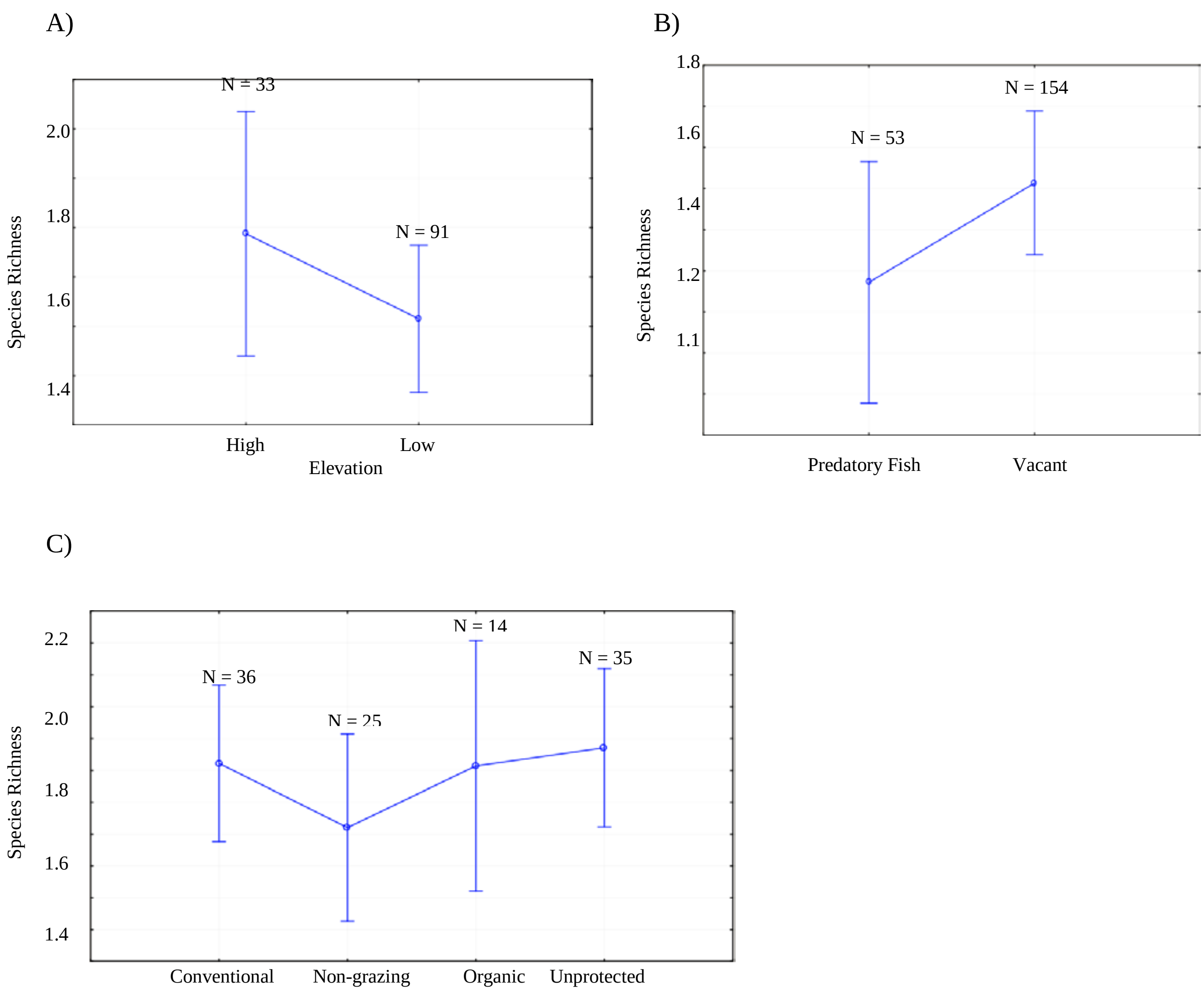
B)


Species
A-PSRE
Skaha Lake
3A
Twin Lakes
White L
Green L
Vaseux Lake
Mahoney L
97
Madden L
Gallagher L
Burnell (Sawmill) L
Tuc-el-Nuit Lake
Spotted L
3
Osoyoos Lake
Blue L
Kilpoola L
97
3
BC
WA
0 1 2 3 4 5 Km
N
S
W
E

C)

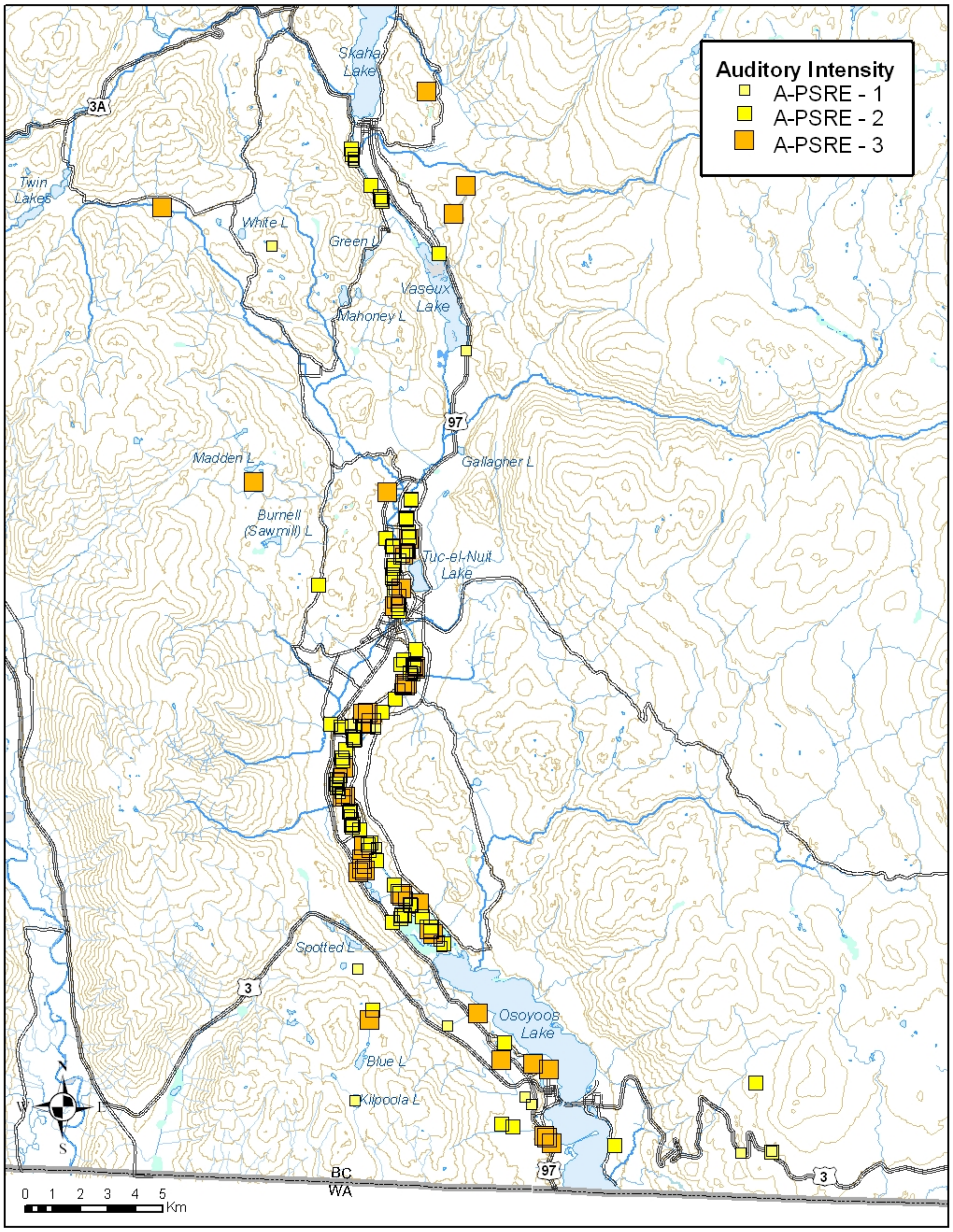

Species
A-SPIN
Skaha Lake
3A
Twin Lakes
White L
Green L
Vaseux Lake
Mahoney L
97
Madden L
Gallagher L
Burnell (Sawmill) L
Tuc-el-Nuit Lake
Spotted L
3
Osoyoos Lake
Blue L
Kilpoola L
BC
WA
97
3
0 1 2 3 4 5
Km

3.7 Species richness and relative reproductive density between sites with fish, by elevation class and land-use subclass, south Okanagan Valley, B.C., 2003 to 2006 (A). When using the whole monitoring dataset species richness did not vary between sites with the presence of fish compared to vacant sites (GLZ: Wald $\chi2$ (1) = 1.41, p = 0.235). (B) When using the relative reproductive density dataset no differences were observed among land-use sub classes (GLZ: Wald $\chi2$ (3) = 1.82, p = 0.610). (C) When using the relative reproductive density data no differences were observed between low and high elevation classes (GLZ: Wald $\chi2$ (1) = 1.37, p = 0.241). Where N represents the total number of samples per site in the analysis. Bars denote 84% confidence intervals.

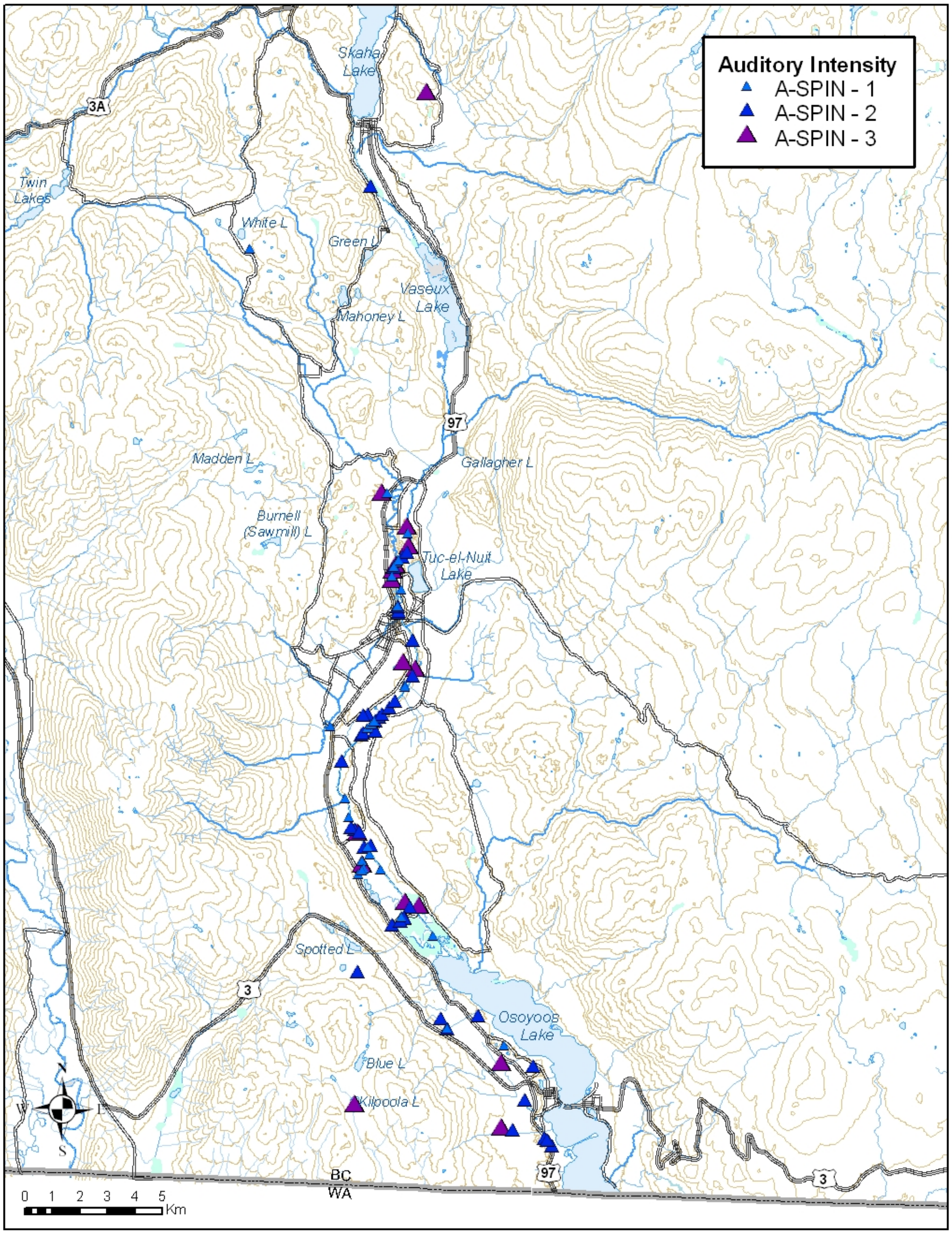


3.8 Auditory intensity calling index for the Pacific chorus frog (A-PSRE) (A) and Great Basin spadefoot (A-SPIN) (B), south Okanagan Valley, B.C., 2003 to 2006. Detected as individuals (index 1), overlapping individuals (index 2) or full choruses (index 3).

A)



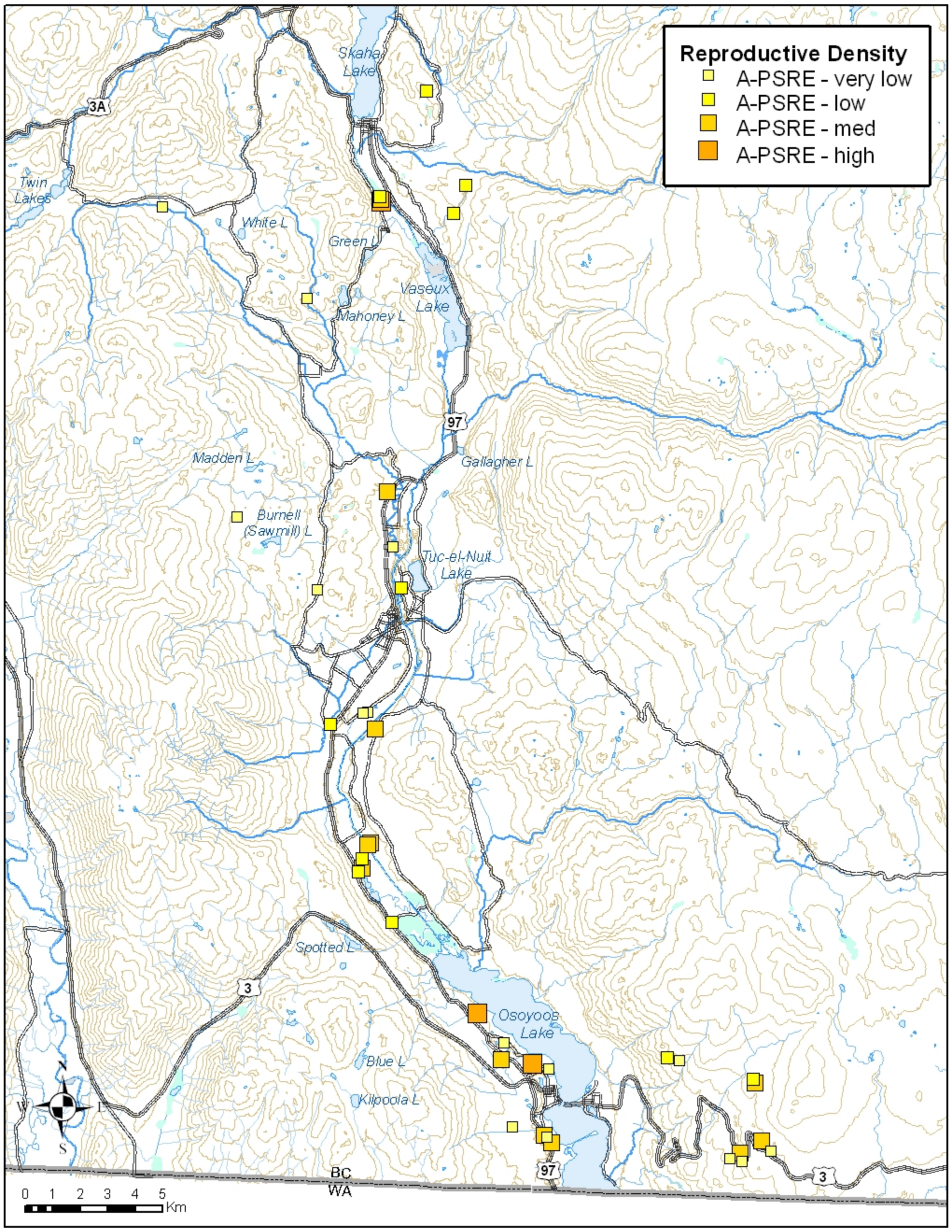
B)



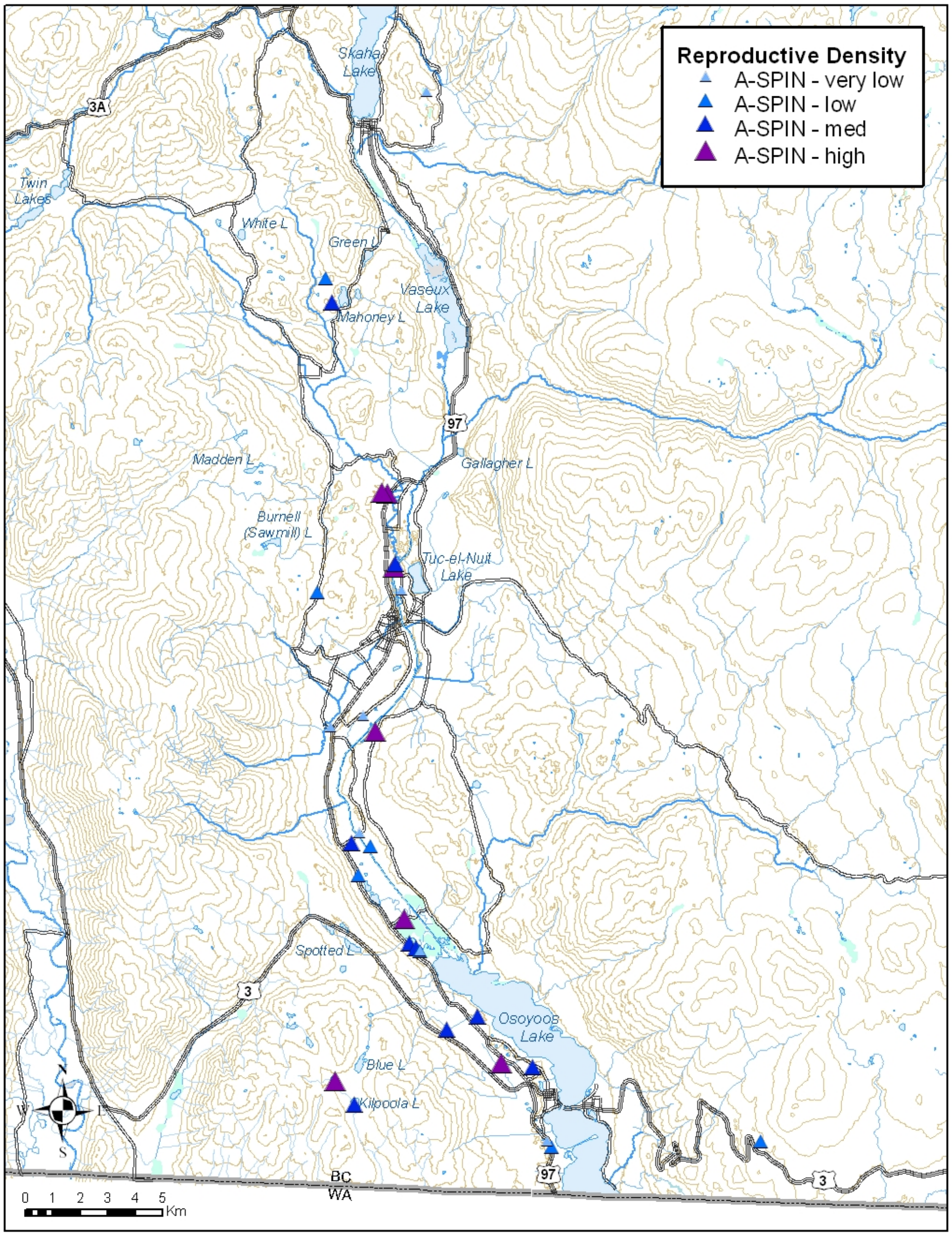
Auditory Intensity
A-SPIN - 1
A-SPIN - 2
A-SPIN - 3
Skaha Lake
Twin Lakes
White L
Green L
Vaseux Lake
Mahoney L
Madden L
Gallagher L
Burnell (Sawmill) L
Tuc-el-Nuit Lake
Spotted L
Osoyoos Lake
Blue L
Kilpoola L
3A
97
3
BC
WA
0 1 2 3 4 5
Km
N
S
W
E

3.9. Pacific chorus frog (A-psre)(A) and Great Basin spadefoot (A-spin) (B) relative reproductive density ($N_{2003\text{-}2006}$ = 43 sites), south Okanagan Valley, B.C. the number of early life stage individuals was categorized as very low = 1 to 9, low = 10 to 99, medium = 100 to 999 or high ≥ 1000 individuals).

A)



B)



Reproductive Density
A-SPIN - very low
A-SPIN - low
A-SPIN - med
A-SPIN - high
Skaha Lake
3A
Twin Lakes
White L
Green L
Vaseux Lake
Mahoney L
97
Madden L
Gallagher L
Burnell (Sawmill) L
Tuc-el-Nuit Lake
Spotted L
3
Osoyoos Lake
Blue L
Kilpoola L
BC
WA
0 1 2 3 4 5 Km

3.10. Relative reproductive density among site subclasses at discrete wetland sites ($N_{2003\text{-}2006}$ = 64 wetland sites analyzed), south Okanagan Valley, B.C.. The number of early life stage individuals was categorized as very low = 1 to 9, low = 10 to 99, medium = 100 to 999 or high ≥ 1000 individuals. For example early life stages of Pacific choris frogs were observed in 5 of 14 Organic Orchard sites over the study period. The relative density frequency is also reported, for example Pacific choris s in Organic Orchards were observed at very low densities on 5 occasions, at low densities on 3 occasions, at medium densities on one occasion, and not observed on any occasion at a high density.

Subclass	Total N_{sites} within subclass	Species	Total N_{sites} species observed at	Relative reproductive density (number represents frequency of density category observed			
				Very low (1 to 9)	Low (10 to 99)	Medium (100 to 999)	High (≥ 1000)
Organic orchard	14	A-PSRE	5	5	3	1	0
		A-SPIN	3	1	1	1	0
		R-PATU	3	3	0	0	0
		A-AMMV	2	2	0	0	0
Conventional orchard	18	A-PSRE	8	6	6	3	1
		A-SPIN	6	5	5	4	2
		R-PATU	6	6	0	0	0
		A-AMMV	3	3	0	0	0
		A-RALU	1	1	1	0	0
		A-LICA*	2	2	1	1	1
Protected grazing	10	A-PSRE	1	1	0	0	0
		A-SPIN	1	1	0	0	0
		R-PATU	2	2	0	0	0
Unprotected grazing	39	A-PSRE	15	11	5	4	0
		A-SPIN	8	4	4	3	0
		R-PATU	4	4	0	0	0
		A-AMMV	9	7	3	2	0
		A-RALU	3		2	0	2
		A-BUBO	3	1	1	0	1
		A-AMMA	7	7	5	0	0
Non-grazing	10	A-PSRE	9	8	6	2	1
		A-SPIN	7	2	3	2	3
		R-PATU	3	3	0	0	0
		A-AMMV	2	2	0	0	0
		A-RALU	2	0	2	0	0
		A-BUBO	1	1	0	0	0
		A-AMMA	2	1	1	0	0
Artificial pool	4	A-PSRE	2	1	2	1	0
		A-SPIN	2	2	1	1	1
Golf course	2	A-PSRE	1	1	0	0	0
		A-AMMV	1	1	0	0	0
Residential	7	A-PSRE	2	2	0	0	0

*American bullfrogs were being actively removed from the two conventional orchard sites (data not presented).

3.11. Mean Pacific chorus frog reproductive densities differed significantly among sub-classes (Wald X^2 (7) = 17.630, p = 0.014), south Okanagan valley, B.C., 2003 to 2006 (A). Similarly, Western painted turtles (GLZ: Wald $\chi 2$ (5) = 17.41, p = 0.004) density was greatest at protected grazing sites (RD 0.202: 8.246 - 1.859, 84% CI)(B). Mean Great Basin spadefoot reproductive densities did not significantly differ among sub classes (Wald X^2 (4) = 8.645, p = 0.07). Where N represents the total number of samples per site in the analysis. Bars denote 84% confidence intervals (C). Highest reproductive densities for a species were observed among Great Basin spadefoots in non-grazing sites (D).

A)

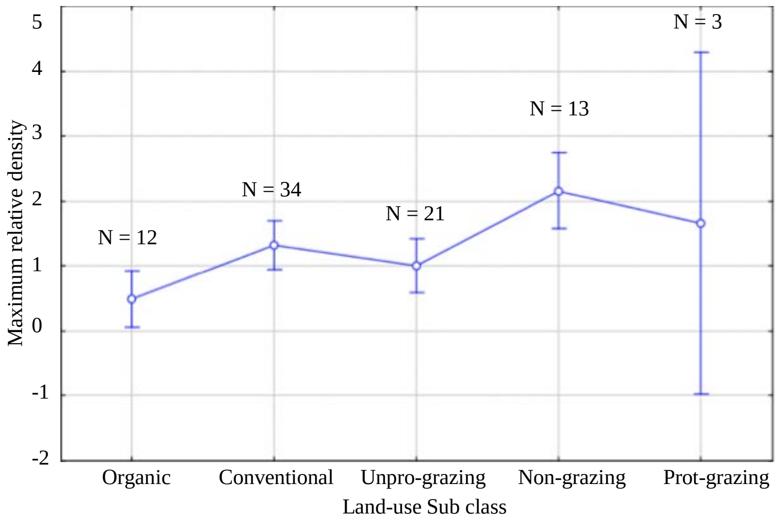
Sub class	**N_{site}**	**Mean**	**Standard Error**	**Max Rel Density -84 CI**	**Max Rel Density + 84 CI**
Organic orchard	9	0.667	0.169	0.423	0.911
Conventional orchard	12	1.590	0.227	1.266	1.913
Unprotected grazing	21	1.184	0.176	0.931	1.437
Non-grazing	14	1.269	0.226	0.942	1.596
Artificial pool	4	1.333	0.494	0.518	2.149
Golf course	1	1.000			
Residential	2	0.400	0.245	-0.022	0.822
Protected grazing	4	0.167	0.167	-0.108	0.442

B)

Sub class	**N_{site}**	**Mean**	**Standard Error**	**Max Rel Density -84 CI**	**Max Rel Density + 84 CI**
Organic orchard		1.000+			
Conventional orchard		1.200	0.92	1.066	1.334
Unprotected grazing		1.100	0.100	0.947	1.253
Non-grazing		1.000+			
Residential		1.000+			
Protected grazing		1.571	0.202	1.248	1.895

+ In some sites, the response is the same for every observation resulting in no observed variance. Due to low sample sizes golf course and artificial sub classes were dropped from analysis

C)



D)

Sub class	**N_{site}**	**Mean**	**Standard Error**	**Max Rel Density -84 CI**	**Max Rel Density + 84 CI**
Organic orchard	12	0.500	0.289	0.065	0.935
Conventional orchard	34	1.324	0.256	0.956	1.691
Unprotected grazing	21	1.000	0.285	0.585	1.415
Non-grazing	13	2.154	0.390	1.570	2.738
Protected grazing	3	1.667	1.202	-0.965	4.298

4.1. Known amphibian and turtle breeding sites within the south Okanagan Valley, B.C. study area. Breeding success was presumed by the presence of early life stages (e.g. eggs, tadpoles, hatchlings, metamorph). Species codes (A Amphibian, R Reptile): AMMV Blotched tiger salamander, AMMA Long-toed salamander, SPIN Great Basin spadefoot, ANBO Western toad, RALU Columbia spotted frog, PSRE Pacific chorus frog, LICA American bullfrog, CHPI Western painted turtle.