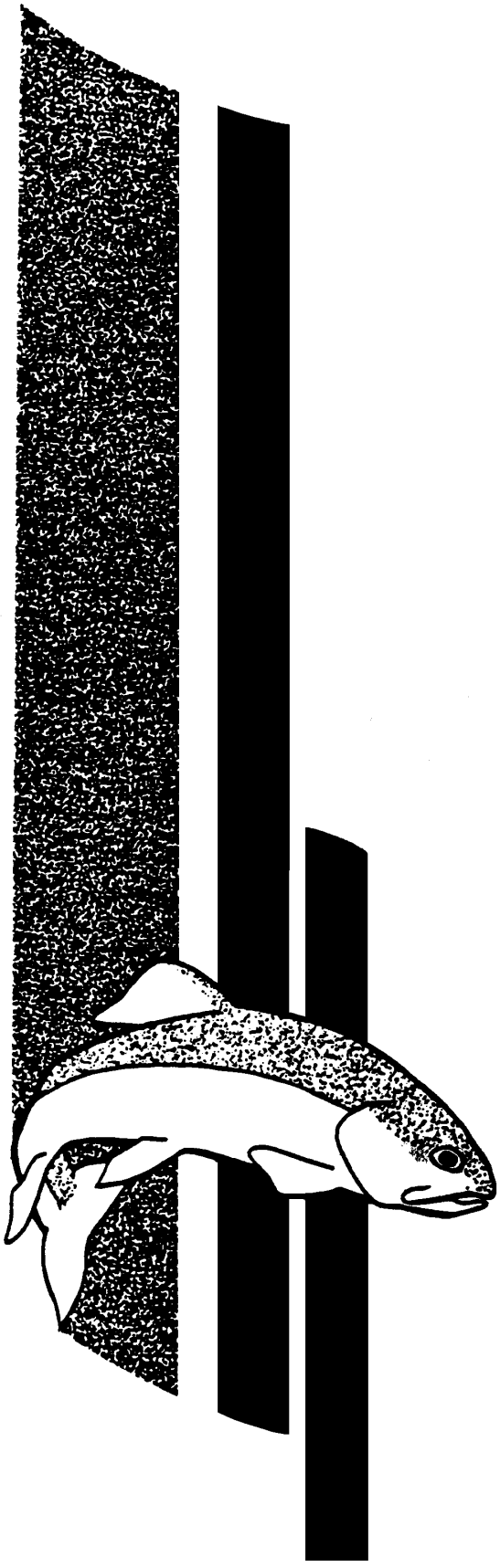


Non-Native Freshwater Fish Species in British Columbia:Biology, Biotic Effects, and Potential Management Actions

by

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Province of British Columbia

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ABSTRACT

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This report summarizes the status of non-native freshwater fish species presence in British Columbia, as well as the state of information regarding biotic effects of these species, up to 2005. A brief overview of invasive theory is provided, including the steps involved from pre-inoculation to impact. The various ecological, social and economic impacts associated with aquatic invasive fish species are described, as well as vectors for introduction. Eighteen fish species non-native to British Columbia or major regions of the province have been reported at least several times based on a data compilation of the provincial fisheries databases that track freshwater fish occurrence information. Four additional species have been observed in a single instance. The biology, habitat needs, potential impacts and recommended management responses are provided for each species in a species account, in addition to a mapped distribution summary. An overall provincial summary regarding trends and suggested management approach is provided.

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Table of Contents

1. INTRODUCTION 1

1.1 PURPOSE OF THIS REPORT 1
1.2 TERMINOLOGY 2

2. ECOLOGICAL PATTERNS AND PROCESSES 3

2.1 STAGES OF INVASION 3
2.2 SPECIES SCREENING AND RISK ASSESSMENT 5
2.3 PREDICTIONS OF INVASIBILITY 6
2.4 GENERALIZATIONS OF THE INVASION PROCESS FOR FISH 8

3. IMPACTS 10

3.1 MEASURES OF BIOTIC CHANGE 10
3.2 BIOLOGICAL IMPACTS 12
3.3 ECONOMIC IMPACTS 15

4. VECTORS OF INTRODUCTION 16

4.1 NATURAL DISPERSAL 16
4.2 ANGLERS AND FISH STOCKING PROGRAMS 17
4.3 RELEASE OF PET, POND AND LIVE MARKET FISH 18
4.4 SHIPPING CONTAMINANTS 19
4.5 AQUACULTURE 19
4.6 BIOLOGICAL CONTROL 20
4.7 WATER DIVERSIONS 20
4.8 BARRIER REMOVAL 21
4.9 SHIPMENT OF LIVE ORGANISMS 21

5. MANAGEMENT OF NON-NATIVE SPECIES 21

5.1 CONTROL 21
5.1.1 Prevention 21
5.1.2 Eradication 25
5.1.3 Maintenance Control 25
5.2 PHYSICAL METHODS OF CONTROL 26
5.3 CHEMICAL METHODS OF CONTROL 30
5.4 BIOLOGICAL CONTROL 31
5.5 DETERMINANTS OF SUCCESS OF CONTROL EFFORTS 32
5.6 SUMMARY 32

6. NON-NATIVE FRESHWATER FISH IN BRITISH COLUMBIA 33

6.1 DATA AND ANALYSES 36
6.2 TRANSLOCATED NATIVE SPECIES 40
6.2.1 Coastal Cutthroat Trout (Oncorhynchus clarki clarki) 41
6.2.2 Westslope Cutthroat Trout (Oncorhynchus clarki lewisi) 42
6.2.3 Rainbow Trout / Steelhead (Oncorhynchus mykiss) 43
6.2.4 Kokanee (Oncorhynchus nerka) 44
6.2.5 Lake Trout (Salvelinus namaycush) 44
6.2.6 Lake Whitefish (Coregonus clupeaformis) 45
6.3 AMERICAN SHAD (*ALOSA SAPIDISSIMA*) 56
6.3.1 Taxonomy 56
6.3.2 Habitat 56
6.3.3 Diet 57
6.3.4 Life History 57
6.3.5 Human Interest 58

6.3.6 Distribution *58*
6.3.7 Threats *60*
6.3.8 Potential Management Actions *60*
6.4 ATLANTIC SALMON (*SALMO SALAR*) 62
6.4.1 Taxonomy *62*
6.4.2 Habitat *62*
6.4.3 Diet *63*
6.4.4 Life History *63*
6.4.5 Human Interest *64*
6.4.6 Distribution *65*
6.4.7 Threats *68*
6.4.8 Potential Management Actions *69*
6.5 BLACK BULLHEAD (*AMEIURUS MELAS*) 70
6.5.1 Taxonomy *70*
6.5.2 Habitat *70*
6.5.3 Diet *70*
6.5.4 Life History *70*
6.5.5 Human Interest *71*
6.5.6 Distribution *71*
6.5.7 Threats *73*
6.5.8 Potential Management Actions *74*
6.6 BLACK CRAPPIE (*POMOXIS NIGROMACULATUS*) 75
6.6.1 Taxonomy *75*
6.6.2 Habitat *75*
6.6.3 Diet *75*
6.6.4 Life History *75*
6.6.5 Human Interest *76*
6.6.6 Distribution *76*
6.6.7 Threats *78*
6.6.8 Potential Management Actions *79*
6.7 BROOK TROUT (*SALVELINUS FONTINALIS*) 80
6.7.1 Taxonomy *80*
6.7.2 Habitat *80*
6.7.3 Life History *81*
6.7.4 Diet *81*
6.7.5 Human Interest *82*
6.7.6 Distribution *82*
6.7.7 Threats *88*
6.7.8 Potential Management Actions *90*
6.8 BROWN BULLHEAD (*AMEIURUS NEBULOSUS*) 93
6.8.1 Taxonomy *93*
6.8.2 Habitat *93*
6.8.3 Diet *94*
6.8.4 Life History *94*
6.8.5 Human Interest *95*
6.8.6 Distribution *95*
6.8.7 Threats *97*
6.8.8 Potential Management Actions *97*
6.9 BROWN TROUT (*SALMO TRUTTA*) 99
6.9.1 Taxonomy *99*
6.9.2 Habitat *99*
6.9.3 Diet *100*
6.9.4 Life History *100*
6.9.5 Human Interest *101*
6.9.6 Distribution *101*
6.9.7 Threats *104*

6.9.8 Potential Management Actions *105*
6.10 COMMON CARP (*CYPRINUS CARPIO*) 107
6.10.1 Taxonomy *107*
6.10.2 Habitat *107*
6.10.3 Diet *108*
6.10.4 Life History *108*
6.10.5 Human Interest *108*
6.10.6 Distribution *109*
6.10.7 Threats *111*
6.10.8 Potential Management Actions *112*
6.11 FATHEAD MINNOW (*PIMEPHALES PROMELAS*) 114
6.11.1 Taxonomy *114*
6.11.2 Habitat *114*
6.11.3 Diet *114*
6.11.4 Life History *114*
6.11.5 Human Interest *115*
6.11.6 Distribution *115*
6.11.7 Threats *118*
6.11.8 Potential Management Actions *118*
6.12 GOLDFISH (*CARASSIUS AURATUS*) 119
6.12.1 Taxonomy *119*
6.12.2 Habitat *119*
6.12.3 Diet *119*
6.12.4 Life History *120*
6.12.5 Human Interest *120*
6.12.6 Distribution *120*
6.12.7 Threats *123*
6.12.8 Potential Management Actions *123*
6.13 LARGEMOUTH BASS (*MICROPTERUS SALMOIDES*) 124
6.13.1 Taxonomy *124*
6.13.2 Habitat *124*
6.13.3 Diet *124*
6.13.4 Life History *125*
6.13.5 Human Interest *125*
6.13.6 Distribution *126*
6.13.7 Threats *128*
6.13.8 Potential Management Actions *129*
6.14 NORTHERN PIKE (*ESOX LUCIUS*) 130
6.14.1 Taxonomy *130*
6.14.2 Habitat *130*
6.14.3 Diet *130*
6.14.4 Life History *131*
6.14.5 Human Interest *131*
6.14.6 Distribution *131*
6.14.7 Threats *133*
6.14.8 Potential Management Actions *135*
6.15 PUMPKINSEED (*LEPOMIS GIBBOSUS*) 136
6.15.1 Taxonomy *136*
6.15.2 Habitat *136*
6.15.3 Diet *136*
6.15.4 Life History *136*
6.15.5 Human Interest *137*
6.15.6 Distribution *137*
6.15.7 Threats *140*
6.15.8 Potential Management Actions *140*
6.16 SMALLMOUTH BASS (*MICROPTERUS DOLOMIEUI*) 142

6.16.1 Taxonomy *142*
6.16.2 Habitat *142*
6.16.3 Diet *143*
6.16.4 Life History *143*
6.16.5 Human Interest *144*
6.16.6 Distribution *144*
6.16.7 Threats *148*
6.16.8 Potential Management Actions *149*
6.17 TENCH (*TINCA TINCA*) 151
6.17.1 Taxonomy *151*
6.17.2 Habitat *151*
6.17.3 Diet *151*
6.17.4 Life History *152*
6.17.5 Human Interest *152*
6.17.6 Distribution *152*
6.17.7 Threats *155*
6.17.8 Potential Management Actions *155*
6.18 WALLEYE (*SANDER VITREUS*) 156
6.18.1 Taxonomy *156*
6.18.2 Habitat *156*
6.18.3 Diet *157*
6.18.4 Life History *157*
6.18.5 Human Interest *158*
6.18.6 Distribution *158*
6.18.7 Threats *160*
6.18.8 Potential Management Actions *160*
YELLOW PERCH (*PERCA FLAVESCENS*) 162
6.18.9 Taxonomy *162*
6.18.10 Habitat *162*
6.18.11 Diet *163*
6.18.12 Life History *163*
6.18.13 Human Interest *163*
6.18.14 Distribution *163*
6.18.15 Threats *165*
6.18.16 Potential Management Actions *166*

7. PROVINCEWIDE PATTERNS AND TRENDS 168

7.1 REGIONAL TRENDS AND PATTERNS 169
7.2 RECOMMENDATIONS 184
7.2.1 Authorized Stocking Programs *184*
7.2.2 Illegal / Unauthorized Introductions *184*
7.2.3 Natural Dispersal from Other Jurisdictions *185*
7.3 CONCLUSIONS 185

8. LITERATURE CITED 187

9. PERSONAL COMMUNICATIONS 206

List of Tables

TABLE 1. THE PRIMARY FRESHWATER FISH SPECIES INTRODUCED TO BRITISH COLUMBIA. 34
TABLE 2. RARE OBSERVATIONS OF NON-NATIVE SPECIES. 36
TABLE 3. BC MINISTRY OF ENVIRONMENT REGIONS. 39
TABLE 4. NUMBER OF NON-NATIVE FRESHWATER FISH SPECIES IN BC IN THREE TIME PERIODS. 168
TABLE 5. SUMMARY OF UNIQUE WATERBODIES IN EACH MOE REGION WITH NON-NATIVE FISH SPECIES.....168

List of Figures

FIGURE 1. INVASION: A MULTI-STEP PROCESS.....3
FIGURE 2. PRE-1976 STOCKING LOCATIONS OF COASTAL CUTTHROAT TROUT IN BRITISH COLUMBIA.....46
FIGURE 3. POST-1975 STOCKING LOCATIONS OF COASTAL CUTTHROAT TROUT IN BRITISH COLUMBIA.....47
FIGURE 4. PRE-1976 STOCKING LOCATIONS OF WESTSLOPE CUTTHROAT TROUT IN BRITISH COLUMBIA. 48
FIGURE 5. POST-1975 STOCKING LOCATIONS OF WESTSLOPE CUTTHROAT TROUT IN BRITISH COLUMBIA. 49
FIGURE 6. PRE-1976 STOCKING LOCATIONS OF RAINBOW AND STEELHEAD TROUT IN BRITISH COLUMBIA. 50
FIGURE 7. POST-1975 STOCKING LOCATIONS OF RAINBOW AND STEELHEAD TROUT IN BRITISH COLUMBIA 51
FIGURE 8. PRE-1976 STOCKING LOCATIONS OF KOKANEE IN BRITISH COLUMBIA 52
FIGURE 9. POST-1975 STOCKING LOCATIONS OF KOKANEE IN BRITISH COLUMBIA.....53
FIGURE 10. STOCKING LOCATIONS OF LAKE TROUT IN BRITISH COLUMBIA.....54
FIGURE 11. STOCKING LOCATIONS OF LAKE WHITEFISH IN BRITISH COLUMBIA 55
FIGURE 12. AMERICAN SHAD OCCURRENCES IN BRITISH COLUMBIA.....61
FIGURE 13. ATLANTIC SALMON OCCURRENCES IN BRITISH COLUMBIA 67
FIGURE 14. BLACK CATFISH OCCURRENCES IN BRITISH COLUMBIA 72
FIGURE 15. BLACK CRAPPIE OCCURRENCES IN BRITISH COLUMBIA.....77
FIGURE 16. BROOK TROUT OCCURRENCES IN BRITISH COLUMBIA.....83
FIGURE 17. PRE-1976 STOCKING LOCATIONS OF BROOK TROUT IN BRITISH COLUMBIA 84
FIGURE 18. POST-1975 STOCKING LOCATIONS OF BROOK TROUT IN BRITISH COLUMBIA 85
FIGURE 19. BROWN CATFISH OCCURRENCES IN BRITISH COLUMBIA 96
FIGURE 20. BROWN TROUT OCCURRENCES IN BRITISH COLUMBIA. 102
FIGURE 21. CARP AND GOLDFISH OCCURRENCES IN BRITISH COLUMBIA.....110
FIGURE 22. FATHEAD MINNOW OCCURRENCES IN BRITISH COLUMBIA. 116
FIGURE 23. CARP AND GOLDFISH OCCURRENCES IN BRITISH COLUMBIA.....122
FIGURE 24. LARGEMOUTH BASS OCCURRENCES IN BRITISH COLUMBIA. 127
FIGURE 25. NORTHERN PIKE OCCURRENCES IN BRITISH COLUMBIA. 134
FIGURE 26. SUNFISH OCCURRENCES IN BRITISH COLUMBIA. 138
FIGURE 27. SMALLMOUTH BASS OCCURRENCES IN BRITISH COLUMBIA. 146
FIGURE 28. TENCH OCCURRENCES IN BRITISH COLUMBIA.....154
FIGURE 29. WALLEYE OCCURRENCES IN BRITISH COLUMBIA.....161
FIGURE 30. YELLOW PERCH OCCURRENCES IN BRITISH COLUMBIA.....167
FIGURE 31. OBSERVATIONS OF NON-NATIVE FISH SPECIES IN BRITISH COLUMBIA PRIOR TO 1950. 172
FIGURE 32. OBSERVATIONS OF NON-NATIVE FISH SPECIES IN BRITISH COLUMBIA PRIOR TO 1976. 173
FIGURE 33. OBSERVATIONS OF NON-NATIVE FISH SPECIES IN BRITISH COLUMBIA 174
FIGURE 34. ALL OBSERVATIONS OF NON-NATIVE FISH SPECIES IN MOE REGION 1 (VANCOUVER ISLAND).....175
FIGURE 35. ALL OBSERVATIONS OF NON-NATIVE FISH SPECIES IN MOE REGION 2 (LOWER MAINLAND). 176
FIGURE 36. ALL OBSERVATIONS OF NON-NATIVE FISH SPECIES IN MOE REGION 3 (THOMPSON).....177
FIGURE 37. ALL OBSERVATIONS OF NON-NATIVE FISH SPECIES IN MOE REGION 4 (KOOTENAY).....178
FIGURE 38. ALL OBSERVATIONS OF NON-NATIVE FISH SPECIES IN MOE REGION 5 (CARIBOO) 179
FIGURE 39. ALL OBSERVATIONS OF NON-NATIVE FISH SPECIES IN MOE REGION 6 (SKEENA) 180
FIGURE 40. ALL OBSERVATIONS OF NON-NATIVE FISH SPECIES IN MOE REGION 7 (WILLISTON).....181
FIGURE 41. ALL OBSERVATIONS OF NON-NATIVE FISH SPECIES IN MOE REGION 8 (OKANAGAN).....182
FIGURE 42. ALL OBSERVATIONS OF NON-NATIVE FISH SPECIES IN MOE REGION 9 (PEACE) 183

1. INTRODUCTION

The ecological effects of non-native species are commonly cited as the second greatest threat to global biodiversity, after habitat loss (e.g., D'Antonio and Vitousek 1992; Wilcove 1998). In freshwater systems, non-native species are cited as the greatest agent of biotic change (Sala et al. 2000). Species are being moved around the globe, both wilfully and accidentally, at an astonishing rate. Vitousek et al. (1997) write that 940 alien vascular plants have become naturalized in Canada, or a little more than 22% of all plant species in the country. British Columbia fares only marginally better at 21%, and even the relatively isolated Queen Charlotte Islands register at almost 20%. Though these numbers do not indicate the geographic range of each species, they are in some ways conservative, since they do not include estimates of translocations within these territories. As many as 50,000 different species have been brought into the US and its territories (Pimentel et al. 2000, 2004) with the number of "successful" introductions continuing to increase (OTA 1993). This is likely much higher than the number of species introduced to Canada, due in part to the tendency for introductions to occur more readily in locales with mild climates and large human populations, like Florida and California (McKinney 2001; Gido et al. 2004). Still, these various estimates indicate the scale of the "onslaught" (Soulé 1990), which is on the order of biological changes wrought from past glaciation-deglaciation cycles, yet occurring over a tiny fraction of the time (Mack et al. 2000). Although some species are deliberately brought into the country, most are merely passengers of global economic trade. It is difficult to regulate these passive imports, so the number of non-native species in Canada and elsewhere is expected to continue to grow.

Effects of these numerous introductions are biotic, physical, aesthetic, spiritual, and economic. Biotic effects range from the imperceptible to extensive. Most non-native species are apparently incorporated into ecological communities with little detectable effect (Moyle and Light 1996a, b), whereas some introductions have widespread, severe impacts such as those caused by Nile perch (*Lates niloticus*) in Lake Victoria (Witte et al. 1992, 2000). Examples of severe physical effects include alteration of estuarine sediment dynamics (e.g., European periwinkle [*Littorina littorea*] Bertness 1984), light penetration in lakes (e.g., water hyacinth [*Eichhornia crassipes*] Lowe et al. 2001) and fire regimes in open forests (e.g., invasive grasses, D'Antonio and Vitousek 1992). Aesthetic and spiritual impacts occur when familiar landscapes are altered and become less natural (e.g., through weed introductions or extinctions of locally important species). Economic impacts are difficult to quantify and represent a large data gap (Mack et al. 2000); but those estimates that have been made are alarmingly large (e.g., Pimentel et al. [2000, 2004] estimate economic damages in the US at approximately $US 120 billion/year). Clearly, the problem and the impacts are enormous, global in breadth and will be difficult to address.

1.1 Purpose of this report

The British Columbia Ministry of Environment is developing a provincial approach for the management of non-native fishes in BC. As one of the first steps in the development of policy and management plans, an up-to-date inventory and evaluation of non-native freshwater fish in BC is required. The purpose of this report is to provide:

1. a general review of non-native species biology (patterns, process, impacts) relevant to freshwater fishes in BC,

2. a review of the biology, distribution, mode of invasion, and impacts of each of the main non-native freshwater fish in BC,
3. a review of prevention, mitigation and restoration measures that can potentially be used in BC, and
4. a summary of the BC situation, highlighting areas of greatest apparent concern.

Although the subject of non-native species has been of interest to ecologists for some time (e.g., Elton 1958) it has been a subject of intensive study only fairly recently. To date there have been relatively few studies directed at geographical regions within Canada, and even fewer within British Columbia; and some taxa and ecosystems are much better studied than others. Where possible, we have concentrated on examples of introductions within or close to BC, and on examples involving aquatic species and ecosystems. In many cases, however, we have had to infer potential ecological processes and impacts from studies in distant systems with different taxa.

1.2 Terminology

Species that are introduced outside their native range are discussed in the scientific literature using many different terms (introduced, alien, exotic, invasive, naturalized, non-native, non-indigenous, etc.) that are inconsistently defined. For example, the term "invasive" species has been used to refer generally to species that become established outside their natural range, and other times to refer to a smaller subset that aggressively expand their range beyond the point of introduction and cause considerable ecological change or economic damage. We prefer to divorce the concept of impact (ecological or socio-economic) from aspects of a species' distribution. We will therefore use only the terms "native" and "non-native" species, which we define as follows.

A **native species** is one that occurs naturally within a particular place; a **non-native species** is one that occurs outside its natural geographic range. A **successfully introduced species** is one that has overcome a natural dispersal barrier and become established in habitat outside its natural range. Any species may be **released** into new habitat, but we generally refer to a species as non-native only when it has established a self-sustaining population. We use the term **failed introduction** to refer to species that have been released into habitat beyond their native range, but have not established a self-sustaining population. Overcoming a natural dispersal barrier may occur through human intervention targeted at a species (e.g., fish stocking, habitat enhancement) or through human actions conducted for reasons entirely unrelated to a species' biology (e.g., interbasin water transfers, ballast water discharge). We do not draw a marked distinction among different geographic sources of introductions. That is, a species found naturally within portions of a watershed or a political boundary, which becomes introduced elsewhere in the watershed or political jurisdiction, is essentially involved in the same process as a species that arrives from a different country or continent. Both involve introductions to habitat beyond a species' natural range, and in most respects these are merely distinctions of scale of transport, which run a continuum from small distances to inter-continental biotic exchange. To help draw a general distinction among species that are naturally found only outside British Columbia, and those that have been moved around within the province, we use the term **translocated native species** for the latter.

2. ECOLOGICAL PATTERNS AND PROCESSES

2.1 Stages of Invasion

The introduction and establishment of a species outside its native range can be conceptualized as a process made up by a series of discrete steps — success at one stage is only possible following success in all previous stages. Williamson and Fitter (1996a, b) were among the first to articulate the process in this way, and others (e.g., Moyle and Light 1996a; Kolar and Lodge 2001; Cohen 2002) have since expanded on this idea. Figure 1 presents the idea in diagram.

The basic concept is that the probability of a species becoming integrated into a community is in most cases very small (Williamson and Fitter 1996a). The number of potential invaders is large, but most are not transported to new locations. Of those picked up and transported, most die in transit. Of those surviving the journey that are released into a new environment, most die before reproducing. Of those that successfully reproduce, many fail to reproduce in sufficient numbers to spread beyond their original point of release. And finally, of those that overcome these obstacles most are integrated into ecological communities without substantial impact; only a relatively small subset cause notable ecological or economic harm. As spectacular as some invasions are, successful introduction to new habitat is a low probability event (Sax and Brown 2000).

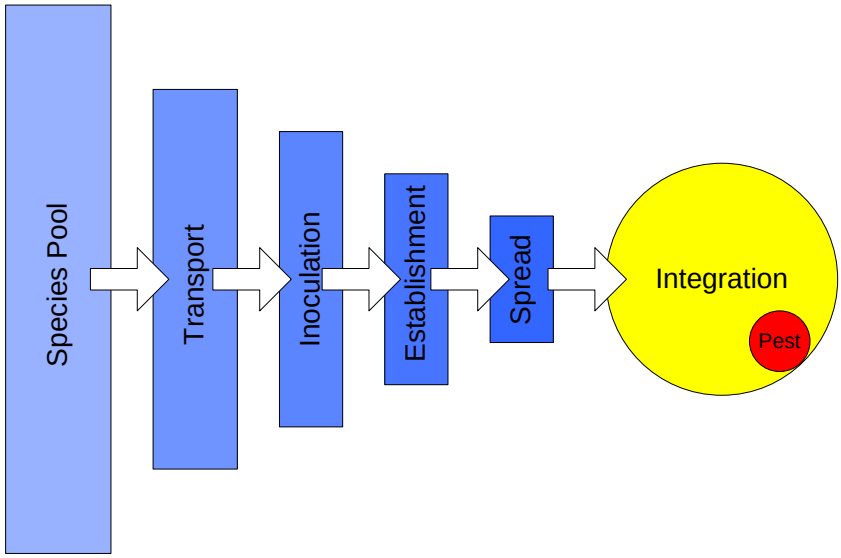


Figure 1. Invasion is a multi-step process, each step of which (usually) has a low probability of success. Each step "filters out" potential invaders, making the cumulative probability of establishment and spread very low. Of those species that do become established, only a subset will become pests.

There are a number of factors that influence an organism's success through each of these stages, including availability, survival through transport, a good physiological and ecological match to the receiving environment, and life history characteristics that favour establishment and spread. Many of these factors are somewhat obvious, others less so. Scientists have spent considerable effort describing and analyzing the factors in an attempt to develop predictive ability for which organisms are likely to invade. For the most part predictive ability is very limited: characteristics that enhance probability of making it through one stage are not necessarily the same as characteristics that influence probability through another stage. There are thus a large

number of trait combinations among successful invaders, making generalizations difficult. The list of traits that apparently make potential invaders successful is bewilderingly large, and often somewhat contradictory (Cohen 2002).

Pre-inoculation.— The first stage of invasion requires that organisms are available for transport and suited to the mode of transport. Both availability and "transportability" are factors that are influenced in part by human values, because values can determine what gets transported and how. For example, fish that are desirable for commercial or recreational fisheries may be numerous and available due to cultivation or collecting activities and transported with care to their new environments. Bass and yellow perch are being actively (and illegally) introduced within some areas of British Columbia, through efforts that require at least some care and attention during transport. Atlantic salmon are not intentionally released in BC, but are made "available" through widespread coastal aquaculture. Other species are introduced through more passive means, such as in ballast water. In this case, probability of successful introduction is drastically increased for organisms with abundant pelagic larvae in source waters.

Inoculation.— Having survived the transport phase a successful invader must be released into a habitat in which it is capable of surviving. Many invasions likely fail at this stage (Moyle and Marchetti 2003) because most organisms are likely introduced to habitat that is not sufficiently suitable for their survival and reproduction. A related factor is the strength of the inoculation, or the number of organisms released, often referred to as "propagule pressure" (Marchetti et al. 2004). Moyle and Marchetti (2003) noted that 70% of fish introductions to California failed when the number of individuals released is less than 100. Some have succeeded with only a handful of fish, whereas others have failed after repeated attempts at introduction (Moyle and Marchetti 2003), however, the general trend is that probability of establishment increases with number of individuals released and number of release events (Marchetti et al. 2004a, b).

Establishment.— Once a species is surviving in its new habitat it must establish a self-supporting population, and ultimately one giving rise to effective dispersers. Moyle and Marchetti (2003) note cases where non-native fish species in California have developed apparently self-supporting populations, but have ultimately failed due to demographic stochasticity or chance event. Marchetti et al. (2004a, b) found statistical support for greatest introduction success among fish with some form of parental care, which they contend helps build up a successful local population.

Spread.— To avoid extinction due to local stochastic events additional populations must ultimately be established. This requires effective dispersal to new habitats, either through natural or human-aided means, effectively creating a loop back to the beginning of the stages indicated in Figure 1. Moyle and Marchetti (2003) note that effective spread to new habitats is often related to a species' desirability to humans.

Impact.— The very possibility of biological invasion presents a paradox (Sax and Brown 2000): how is it that organisms (often from distant locations and with no evolutionary history with local biotic and abiotic conditions) are able to establish themselves and at times even displace native species? One of the main reasons, it seems, is that many ecological communities are not saturated with species (Caley and Schluter 1997; Cohen 2002). In truth, most non-native species are incorporated into local communities with little or no discernable effect (Moyle and Light

1996a). A few cause substantial ecological or economic harm (Moyle and Light 1996a; Pimentel et al. 2004).

The invasion process is complex and making predictions about invasion success has proved to be imprecise at best (Moyle and Light 1996a). There is even greater uncertainty in predicting which species will become pests. It remains a major challenge to understand where the threats are and how best to manage them. The following sections review some of the attempts to develop a predictive science of invasion biology.

2.2 Species Screening and Risk Assessment

Biologists have expended considerable effort analyzing life history traits of non-native species in order to describe the properties that give rise to successful invaders. The effort has often been justified by the notion that if we knew which traits made a successful invader we would be able to screen species for those traits and assess the risk they would impose if imported and released. Although true, this justification fails to acknowledge that many introductions are unintentional, resulting as a by-product of global commerce, from clandestine efforts or by individuals that are ignorant of current regulations. Managing and preventing these introductions is not at all straightforward and risk assessments may not provide much ammunition in the fight.

One of the main problems with efforts to produce species screening tools is that the screening results have been of limited utility due to their generality. Marchetti et al. (2004a, b) defined seven predictor variables (trophic status, size of native range, parental care, maximum adult size, physiological tolerance, distance from nearest native source, and propagule pressure) and evaluated the statistical power of these to predict successful invasion of non-native fish to California watersheds. They found that all seven variables were useful in explaining one or more stages in a successful invasion. Notably, the predictive relationships were different for each of three invasion stages investigated. Each successful invader may have different trait combinations, such that one trait is important for one invasion stage, but not necessarily for another. In this sense, different traits can compensate for one another such that a successful "invasion strategy" can be built in several ways. It is difficult to use such information to assess a priori which species are likely to be successful invaders, and just as importantly which species are not. The information is also difficult to use when assessing which of the species may have some impact on the receiving community.

Kolar and Lodge (2002) used a different statistical methodology to assess the invasion risk of fish to the Great Lakes. The model was built using species that had previously invaded, and was applied to a new set of species that could arrive via ballast water exchange. The method was capable of discriminating among successful invaders the majority of the time, but still had substantial errors in recognizing previously introduced pest species. The model was nevertheless deemed useful for screening a series of fish species that may yet arrive in ballast water. One weakness of the approach is that it is difficult to know a priori the number of false positives (species incorrectly flagged as a "nuisance") produced by this approach.

The utility of risk assessments is not always clear. For example, conducting a detailed screening on a long list of species that may be released from ballast water seems a somewhat moot point.

We already know, based on experience, that ballast water releases are potentially harmful, and restrictions would be more effective if directed to ballast water management rather than to management of the individual species in the ballast water. A second criticism of broad screening efforts is that important details may get lost in the generalities. For example, brown bullhead have been widely introduced in North America and have not caused widespread ecological damage. Yet in BC, introduction of this species is responsible for the extinction of a pair of stickleback species (Hatfield 2001). In other words, the effect of brown bullhead has been relatively minor in most cases, but has been catastrophic in this one instance. Whether this species is classified as a serious concern depends in part on one's point of view.

We see three clear uses of risk assessments. First, the method can be used to help focus control efforts. Selected species could be screened where those species have a high probability of intentional introduction. For example, a risk assessment for pike or bass would be part of a strategy to see what kind of response would be appropriate if the species were found in a new habitat. Second, risk assessments should be done in the case of intentional introductions to new habitat, especially when species are released as biocontrol agents (Simberloff and Stiling 1996). Third, non-native species should be screened when they are brought in for the ornamental or food trade (e.g., horticulture, aquarium trade, aquaculture). A risk assessment will help define a priori whether a species has an acceptable risk of escape and invasion. A difficulty in all these cases will be defining what is "acceptable" risk, which is driven largely by human values that may differ among individuals.

A risk assessment is conducted to some extent in BC when individuals, agencies and businesses apply to the inter-agency Introductions and Transplants Committee (Fisheries and Oceans and Government of British Columbia) for a permit to move and release live aquatic organisms in BC waters including natural waterbodies, ponds, culture facilities and other artificial holding areas. Furthermore, a Prohibited Import of Live Fish list has been developed for the province under the Pacific Fisheries Regulations (Schedule VIII, Section 5) of the federal Fisheries Act (1985). Unfortunately, this list only deals with importation; several of the species found on this list continue to spread in BC. This list is considered by many to be outdated and limited in scope.

2.3 Predictions of Invasibility

What types of habitats are most easily invaded? Biological invasion requires a match between organism and receiving environment, so it is important to review the environments and not just the invaders to understand the mechanisms of biological invasion. One idea is that disturbed habitats are more easily invaded than stable ones. There are good examples of invasions that coincide with (and perhaps depend on) increased or decreased habitat disturbance. Altered fire regimes (either increased fire incidence or fire suppression, depending on the species of invader) are correlated with invasion of non-native grasses in Australia and North and South America (D'Antonio and Vitousek 1992; MacDougall and Turkington 2005). There are numerous examples of fish invasions in disturbed habitats (see discussions in Section 6), but there are also many examples of invasions into relatively undisturbed habitats such as parks (Donald 1987). In a survey of the literature, Lozon and MacIsaac (1997) found that cases of successful biological invasion were approximately equally split between those correlated with disturbance and those that were not. Invasion of plants were commonly associated with animal activities (e.g., grazing, seed introduction), soil disturbance, forestry, fire, agriculture, and

human activities. Establishment of exotic animals appeared to be much less dependent on disturbance. In an examination of rainbow trout introductions worldwide, Fausch et al. (2001) found that successful invasion was best predicted by a match between the hydrologic timing of flooding and emergence time of fry. Introductions were most successful when receiving habitats had disturbance regimes (e.g., timing and magnitude of flood events) similar to those within the species' native range. Disturbance may enhance some invasions, and be required for a few, but it is not an essential component of successful introductions.

What types of communities are most easily invaded? As noted earlier, it seems paradoxical that non-native organisms from distant locations and with no opportunity to adapt to the local environment are able to become established and sometimes to displace native species, which have had a long period of history in which to adapt to local conditions (Sax and Brown 2000). On closer examination this need not be a paradox. All ecological communities are built through a process of species additions and deletions, through both speciation and biotic mixing. In essence, the invasion of non-native species is similar to natural processes of community assembly, it is simply occurring much more rapidly due to a variety of human influences. In fact, the evidence indicates that most introductions do fail (Williamson and Fitter 1996b; Mack et al. 2000), and success is often dependent on repeated releases of large numbers of individuals that are pre-adapted to the new habitat.

The second element of an explanation is that communities are not usually saturated with species. Early explorations of this issue suggested that species-poor communities would be less resistant to invasion than ecological with high diversity. The rationale was that low diversity communities had vacant niches, whereas diverse communities had less available niche space and more intense competition for resources. Since islands are often relatively species poor, the hypothesis can be tested through examination of biotic invasions on islands. There is little empirical support for this hypothesis (Mack et al. 2000). Simberloff (1995) notes that if islands have lower resistance to invasion, it is unlikely related to low diversity, but rather the complete absence of some groups (e.g., terrestrial mammals on some islands). Simberloff also notes that "virtually every kind of damage wrought by invaders on islands has also been wrought in mainland areas." In an examination of bird invasions on islands, Sol (2000) found no evidence that islands are less resistant than continents to invasions. Moyle and Light (1996a) note that in the case of fish invasions in California lakes, streams and estuaries (which are functionally equivalent to islands), successful invasion is usually highly idiosyncratic and there is no apparent extra resistance offered by high diversity. There is no apparent support for the notion that real ecological communities are saturated with species, even those that have high inherent diversity (e.g., Caley and Schluter 1997).

A third element of the explanation is the idea of "enemy release." Many invaders arrive in new habitats without the predators and parasites that occur in their native habitats. They are thus released from their usual biotic constraints and often flourish at rates that exceed those in their native habitats. This explanation is one of the simplest of how a non-native species can thrive in a habitat in which it is not native (Mack et al. 2000). This mechanism has not been widely tested, but the few examples are compelling. For example, in comparison to its native habitat in Australia, the bushtail possum (*Trichosurus vulpecula*) occurs at 10 times the density in its non-native habitat in New Zealand, where it has many fewer competitors and parasites (Mack et al. 2000). Likewise, the plants *Chrysanthemoides monilifera* and *Acacia longifolia* are substantially

fitter (fruiting, seed production) in their non-native habitat than in their native habitats due to the absence of attacks by pests in their new habitats (Mack et al. 2000). Keane and Crawley (2002) review the general concept and a variety of empirical studies and find less than universal support for the hypothesis.

There have been a variety of efforts to explore invaded communities from the perspective of classical community ecology. This perspective views communities as the result of additions and deletions with predation and competition as the primary structuring forces. In reviewing the theoretical and empirical results of community assembly rules, Moyle and Light (1996a) break down the ideas as follows:

1. invasion success decreases with species richness and the amount of time the community has been accumulating species;
2. there is a priority effect for residents: invasion, even by a superior competitor, is difficult at low numbers of invaders;
3. invasion resistance increases as competition and predation become stronger;
4. the final community state depends on invasion order, with multiple outcomes possible from a single species pool.

Although these conclusions were supported by many theoretical studies (e.g., Case 1991) and some experimental studies (e.g., Fargione et al. 2003) there are also many examples where these results do not hold true. For example, Moyle and Light noted that for invading fishes in California streams and estuaries, results suggest that if abiotic conditions are appropriate for an invader, it is likely to become established, regardless of the biota already present. Conversely, invasion failures are best attributed to poor pre-adaptation to abiotic conditions rather than to properties of the recipient community (Baltz and Moyle 1993; Moyle and Light 1996a).

2.4 Generalizations of the Invasion Process for Fish

Williamson and Fitter (1996b) were among the first to divide the invasion process into a series of discrete steps, which they noted each had a low probability of success (see Section 2.1). They divided the process into 3 steps: escaping, establishing and becoming a pest. They reviewed studies from a variety of taxa and assigned probabilities for each to advance to the next stage in the invasion process. Their analysis indicated that on average there was about a 10% chance of success for each step, and they called this the "tens rule." This of course translates to a 0.1 % chance for any species to become introduced, spread and become a pest. Though this work has never been embraced as an accurate reflection of introduction probabilities for all taxa, it helps underscore the notion that the probability of a species becoming integrated into a community and causing widespread impact is in most cases very small.

Moyle and Light (1996a) reviewed patterns of fish introductions in California and attempted to develop generalizations from these. Although they were successful in developing several high level generalizations, they were not successful in developing the kind of precise predictions that would be considered extremely useful for resource managers. One of their primary findings was that the invasion process is highly idiosyncratic: there is no universal set of characters that will predict success or failure of fish invasions or which fishes are likely to become nuisance

species. They developed 12 general rules, which we summarize here because they remain the state of the art for fish introductions and their effect on aquatic communities.

1. Most invaders fail to become established. Direct information on this is limited, but the rule is supported by a wide array of coincidental evidence. A successful introduction requires release of sufficient numbers at the right time and place. Unintentional releases are unlikely to be at the time and place where the receiving environment is available for colonization. Successful intentional introductions, however, tend to rest on an understanding of this relationship.
2. Most successful invaders are integrated without major negative effects (e.g., extirpations) on the communities being invaded. Extirpations of native species do occur following invasions by non-natives, but they are rare, and a relatively small number of species are responsible for most extirpations. Non-native predators are usually responsible for the extirpations.
3. All aquatic systems are invasible and invasibility is not related to diversity of the resident organisms. Fish have been successfully introduced to fishless habitats and highly diverse habitats. There is no evidence that diverse fish communities are more resistant to invasion than low-diversity communities.
4. Major community effects of invasions are most often observed where the number of species is low. There are two apparent reasons for this. The first is that effects are simply more visible in less diverse systems. The second is that diverse systems tend to have had a longer history of additions and deletions and are already the result repeated invasions and community adjustment.
5. In systems that have been minimally altered by human activity, fishes most likely to be successful invaders are top predators and omnivore/detritivores. A review of successfully introduced fish species indicates that predators and omnivores are more likely to become established than fish in mid-trophic levels. The success of these two trophic groups is thought to be related to the high availability of food during the establishment phase of the invasion.
6. Piscivorous invaders are most likely to alter the fish assemblages they invade while omnivore and detritivores are least likely to do so. Piscivores often have a dramatic effect on fish assemblages in lakes and streams, whereas omnivores and detritivores do not. These patterns have been well documented.
7. In aquatic systems with intermediate levels of human disturbance, any species with the right physiological and morphological characteristics can become established. Habitats with intermediate levels of disturbance appear to facilitate the coexistence of a wide variety of species.
8. In the long term, or in relatively undisturbed aquatic systems, success of an invader will depend on a close match between its physiological and life history requirements and the characteristics of the system being invaded. Organisms need to be "pre-adapted" to their new environment. Translocations from nearby areas are more likely to be successful than those from more distant locations because nearby areas are more likely to be similar in physical attributes.
9. Invaders of natural aquatic systems are most likely to become established when native assemblages of organisms have been temporarily disrupted or depleted. Successful invasion appears to be related in part to disruption of existing communities by large human-induced or natural disturbances. This implies that biotic resistance to invasion

into freshwater systems does exist to a certain extent, particularly during the early stages of invasion, and if disrupted by a disturbance increases the chance of successful invasion.

10. Long-term success (integration) of an invading species is much more likely in an aquatic system permanently altered by human activity than in a lightly disturbed system. There are likely a number of reasons for this, but most prominent is that human-disturbed systems (e.g., reservoirs) have many similarities among them and therefore favour a similar suite of species.
11. The invasibility of a natural aquatic system is related to the interactions among environmental variability, predictability, and severity. Poff and Ward (1989) and Poff and Allan (1995) found that composition of fish communities in streams is determined by the broad hydrologic features of watersheds (e.g., annual freshet). Invaders seem to do poorly in highly variable but predictable environments.
12. Invaders are most likely to extirpate native species in aquatic systems with extremely high or extremely low environmental variability or severity. Aquatic systems with benign physical and chemical characteristics are likely to support diverse communities, including both native and non-native species. Declines of native fishes is usually related to both non-native species and major habitat changes. For example, when streams in the western United States are dammed, creating reservoirs and damping the natural fluctuations in stream flow, they are quickly invaded by non-native fishes, to the detriment of the native species.

In summary, it is key that we acknowledge the idiosyncratic nature of invasions when trying to predict how species may affect BC's environment. We can learn from experiences in other jurisdictions in terms of which species have proven most harmful to native biotic community and under what conditions, but at the same time we must recognize that BC waters present a novel set of conditions (and species) that may prove more or less susceptible to invasion.

3. IMPACTS

Non-native species can cause a variety of impacts, including social, economic and environmental. Some of these impacts are severe, whereas many others are subtle and difficult to detect. This section reviews some of the measures of impact, the mechanisms by which non-native species cause impacts, and discusses examples of biological and economic impacts and trends.

3.1 Measures of Biotic Change

Non-native species are often cited as an important driver of global ecological change, and are usually ranked immediately after habitat loss in terms of overall effect. Wilcove et al. (1998) quantified the extent to which various human activities are imperilling plant and animal species in the United States. They examined listings of endangered plant and animal species under the US Endangered Species Act as well as those listed as imperilled by the Nature Conservancy and tallied the reasons for endangerment. Threats were classified into five groups: habitat destruction and degradation, alien species, pollution, overexploitation and disease. Some species are threatened by more than one of these. Habitat loss was implicated in approximately 85% of cases; alien species were ranked as the second leading cause, and were listed as an

important factor in 49% of cases. The other factors were considerably less general: ~25% for pollution, ~20% for overexploitation, and < 5% for disease.

There is broad concurrence that non-native species are causing dramatic changes in ecological systems worldwide, but there is often uncritical acceptance that effects of non-native species are universally severe and negative, and that the effects are constant across all ecosystems. Indeed, some have stated a fear that the scientific literature has become burdened with uncritical generalizations about non-native species (Gurevitch and Padilla 2004; Didham 2005). Gurevitch and Padilla (2004) review several examples of non-native species and their effects on native flora and fauna. They suggest that a closer inspection of many examples shows that impact pathways are not always clear, and there is less evidence of severe impact from non-native species than often reported. In part this is no doubt exacerbated by the fact that causes of species declines are often multifactorial (Didham 2005). The benefit of this work is not to discredit the notion that non-native species can and do have severe impacts, but to focus the discussion on specific causes of impact rather than on non-native species as a group. For example, it may be of greater benefit to address spread of a particular species than to dilute the effort by attempting control of many species, some of which are relatively benign. The discussion also notes that it may be necessary to focus on other drivers of ecological change, such as habitat loss and degradation.

Species richness is decreasing globally (as species go extinct), but increasing at a regional scale (as other species are translocated to new habitats). This process causes a loss of regional distinctness, or a homogenization of global fauna, and has been quantified for several taxonomic groups. Rahel (2000, 2002) quantified the homogenization of fish faunas in the continental US. He noted that loss of regional distinctness can be caused by loss of geographically restricted endemic species and by spread of cosmopolitan species to new habitats. By comparing fish faunas today with faunal distribution prior to European settlement, Rahel noted a substantial homogenization, with introductions playing a much larger role than extirpations in homogenizing fish faunas. He compared pairs of states, and found that 89 pairs that formerly had no species in common, today share an average of 25.2 species. On average pairs of states have 15.4 more species in common today than before European settlement. The vast majority of translocations occurred as a result of attempts to enhance food and sport fisheries, and most of these were widespread efforts, particularly translocations of eastern US fish species into the western US.

Taylor (2004) performed a similar analysis for Canada, with a specific emphasis on British Columbia. In a broad sense, the patterns of homogenization were similar to those in the US discussed by Rahel. However, there was generally less homogenization and greater distinctness among provinces than noted in Rahel's study. Within BC, some ecoregions had become significantly more homogeneous since European settlement, notably Vancouver Island and the Columbia River ecoregion. In our analysis (see Section 7.1) we also note that Vancouver Island and Kootenays are the Ministry of Environment management regions that have the most records of non-native species. The fact that Canada has lower homogenization rates than the US indicates that efforts to prevent species translocations is effort well spent if preservation of regionally distinct fauna is an important goal.

3.2 Biological Impacts

Non-native species cause a range of biological impacts, from subtle to spectacular. Types of biological impact include extinction, altered abundance and distribution, altered food webs, changes in community interactions through species additions and deletions, and altered evolutionary trajectories. These impacts also vary among ecosystems. For example, Sala et al. (2000) suggest that non-native species are the primary driver of biotic change in freshwater systems, whereas land use is the primary driver for terrestrial systems. Extinction and abundance changes are the best-studied impacts of non-native species. Even in these there is variation in the scale of effect. For example, extinction can be at the local, regional, or global scale (Carlton et al. 1999), and numerical decline may result in "functional extinction" if the decline means that a species no longer has a major role in the ecological community (Carlton et al. 1999; Mooney and Cleland 2001).

In this section we briefly review the mechanisms of impact and discuss examples from British Columbia and elsewhere in the world. The review of examples is meant to illustrate the wide array of possible impacts, and is not comprehensive. More detailed reviews are available in Lowe et al. (2001) and the Global Invasive Species Database available at http://www.issg.org/database/welcome/ and throughout the broad literature base on this topic.

Predation.— Non-native predators cause the most consistently negative and measurable impacts in aquatic systems (Moyle and Light 1996a). Predators do not always lead to extinction of other taxa, but their effects tend to be larger and more noticeable than those of introduced planktivores, benthivores or omnivores (Moyle and Light 1996a). We have noticed that exotic species management plans in many jurisdictions concentrate on predatory species (e.g., walleye and pike), however, our review of such plans has not been exhaustive.

Probably the most spectacular example of biotic impact from an non-native fish predator is that of the Nile perch (*Lates niloticus*). Lake Victoria, the world's largest tropical lake, had a fish fauna dominated by more than 500 species of haplochromine cichlids (Greenwood 1981). The Nile perch was introduced into the lake in the 1950s, and its population rapidly increased in the 1980s (Witte et al. 2000). At the same time 200 to 300 (or more) haplochromine cichlid species vanished (Witte et al. 1992).

There are many examples of the effects of non-native predators, though most are less spectacular than the Nile perch example. Non-native predators have played a role in extinction of several fish fauna in the US and Canada, including several species of cisco, dace, chub and pupfish (Miller et al. 1989). Introduced smallmouth bass have caused widespread local extinction of small-bodied littoral fish species in Ontario and Quebec (Chapleau et al. 1997; Whittier et al. 1997; Vander Zanden et al. 1999; Whittier and Kincaid 1999; Findlay et al. 2000; MacRae and Jackson 2001). Introduced brown trout have caused the local extinction of several fish species in New Zealand, primarily through predation (Flecker and Townsend 1994; McDowall 2003; McIntosh et al. 1994; Townsend 1996). Widespread introductions of mosquitofish have had severe effects on native species, primarily through predation on fish eggs and larvae (Meffe et al. 1983). In BC, the introduction of brown bullhead caused the extinction of a stickleback species pair in Hadley Lake, Lasqueti Island, with the mechanism being nest predation (Hatfield 2001). Large impacts are not restricted to aquatic predator

species, of course, as exemplified by the brown tree snake in Guam, mongoose in the Caribbean and South Pacific, feral cats and rats, bullfrog, and others (Lowe et al. 2001).

Competition.— Species richness is decreasing globally but increasing at a regional scale (e.g., Rahel 2000; Mooney and Cleland 2001; Sax et al. 2002; Davis 2003; Taylor 2004). This occurs because species become extinct, for a variety of reasons, leading to a declining number of species on the globe. At the same time, there is ongoing spread of non-native species and incorporation into local ecological communities. In essence, this pattern can only occur if communities are not saturated with species and competition is insufficient to exclude invading taxa. Evidence indicates that communities are not saturated with species (e.g., Caley and Schluter 1997; Loreau 2000). There is also evidence that effects of competition from non-native species are considerably less than those for introduced predators: predation far outweighs competition as a primary cause of extinction over multiple taxa (e.g., Case and Bolger 1991; Mooney and Cleland 2001; Sax et al. 2002; Davis 2003). Competition-driven extinctions have been rare, even on oceanic islands where non-native species have had severe effects (Davis 2003).

In general, interspecific competition is difficult to measure and has led to conflicting conclusions on its importance in determining abundance and distribution of organisms (Connell 1983; Schoener 1983), and assessing the role of competition has occupied a generation of community ecologists. Although competition is difficult to study, and generally less severe in its effects than predation, it is nevertheless a substantial force in the evolution of fish (Robinson and Wilson 1994). There have been concerns raised about almost all the non-native fish species reviewed in Section 6 regarding the potential effects of competition with native species, and it seems reasonable to be concerned about the competitive effects on local taxa. Moyle et al. (1986) discuss fish introductions in California and provide several examples of competitive exclusion as the most likely mechanism for displacement of native fish species. Although there is debate over the relative importance of different mechanisms of spread of brook trout (Dunham et al. 2002), it is likely that competition is a major force in its displacement of native species (Levin et al. 2002; Peterson and Fausch 2003a).

The effect of hatchery-reared stocks on wild populations is a special case of competitive effects of non-native species. Intraspecific competition is more intense than interspecific competition, and the effects of hatchery releases on wild stocks have been widely noted (Weber and Fausch 2005). Where stocking programs supplement natural production competition may be strong enough to result in partial or complete exclusion of native stocks.

Parasitism and Disease.— Few fish species can be classified as parasites, with the exception of some lampreys, and even these are often referred to as predators. Introduction of the sea lamprey into the Great Lakes has had many serious impacts on local fish stocks, and large sums are spent annually on direct control (Environment Canada 2004).

More likely introductions in BC are disease organisms, which can have a significant biotic impact on native species. Such organisms may be 'hitch-hikers' with other introduced species or spread unaided. Most of these are micro-organisms and often difficult to control once released. Some examples of non-native disease organisms causing widespread ecological harm are *Myxobolus cerebralis* (the causative agent of whirling disease, which has become widely

established in wild salmonid populations in North America), *Aphanomyces astaci* (a macro-fungus that attacks European native crayfish introduced from North America), *Batrachochytrium dendrobatidis* (a macro-fungus linked to amphibian die-offs in Panama and Australia), West Nile virus (transmitted from birds to humans), and *Plasmodium relictum* (the mosquito-transmitted avian malarial parasite, which is lethal to some species). The best defence against many of the disease organisms is to avoid translocation of organisms. Non-native disease organisms are of particular concern because native species have not evolved natural resistance to the diseases.

Hybridization.— One tends to think of predation and competition by non-native species as the largest and most frequent impact mechanisms. This is true in most cases, however, in some instances these effects may be dwarfed by other factors such as hybridization. Hybridization occurs when individuals of different taxa (genera, species, population, etc.) mate. The use of molecular genetic techniques has shown a higher incidence of hybridization in endangered taxa than originally detected by morphological studies (Rhymer and Simberloff 1996). Introgression (gene flow) may occur if reproductively viable offspring are produced, but even when they are not produced (i.e., offspring are sterile, or not produced at all), there may be concerns over wasted mating effort. Rhymer and Simberloff (1996) review a number of cases and show that hybridization is a conservation concern in a wide variety of taxa, though it is most common in plants and freshwater fish. Most often hybridization is a possibility when taxa are released from relatively nearby, rather than distant locales (Perry et al. 2002).

Hybridization can result from direct introduction of a taxonomically similar species, after habitat modification brings related species into sympatry, or after ecological change severely numerically reduces relative abundance of a species. The latter two are somewhat indirect results of non-native species. In one instance, contact barriers are overcome through modification of the barrier (either through human actions or through activities of a non-native species), rather than direct transportation and release of organisms. In the other instance a non-native species may numerically impact one or more species (e.g., through predation) and thereby increase hybridization: cross-matings often increase when taxa become rare and may be proportionately asymmetrical when one species is rare and the other common (Rhymer and Simberloff 1996).

A well-known example of hybridization following introduction of a non-native species is that of eastern brook trout introduced to areas with bull trout. These two species are members of the same genus, *Salvelinus*, and frequently hybridize in sympatry (Kanda et al. 2002). Matings appear to be at random, at least in some locations (Leary et al. 1993). There appears to be little or no gene flow from matings between bull trout and brook trout, although viable F1 hybrid offspring are produced (Kanda et al. 2002). Brook trout also compete with bull trout, but hybridization is believed to facilitate displacement because reproductive effort is wasted, particularly in the numerically less abundant bull trout (Leary et al. 1993).

Miller et al. (1989) reviewed case histories of recently extinct fishes in North America, a total of 40 taxa. The cause of extinction was often difficult to determine, but hybridization was deemed to be a factor in 38% of the cases. The mechanism of impact from non-native species was usually indirect, following numerical decline as a result of introduced predators or habitat change (Miller et al. 1989).

A special case of hybridization effects occurs as a result of fish stocking programs for enhancement of sport fishing. There are gradations to this impact: in some instances stocked fish interbreed with different taxa (e.g., introduced rainbow interbreeding with native Apache and Gila trout in western US [Dowling and Childs 1992]), and in other instances "interbreeding" occurs within the same species between naturally produced individuals and those of hatchery origin. There are numerous and substantial concerns with the genetic consequences of augmenting natural production with hatchery production (Allendorf and Waples 1996).

Habitat Modification.— Through the everyday activities of growing, feeding, and reproducing, plants and animals modify their surroundings. Sometimes these modifications can have consequences for other organisms. Some modifications are large and severe, others much less so.

The non-native zebra mussel has had an enormous biotic and economic impact on Great Lakes and surrounding aquatic ecosystems, converting large areas of open benthic substrates to dense mats of mussels. Zebra mussels have displaced native benthic organisms and altered aquatic food webs (Raikow 2004) and had enormous economic impact through fouling of physical structures (Pimentel et al. 2000). Some plant species act in similar ways, Eurasian milfoil in shallow lakes, *Caulerpa* seaweed in the Mediterranean, Miconia trees in Tahiti, water hyacinth in many tropical and subtropical areas (Lowe et al. 2001). Other species are less spectacular in their local abundance and distribution, but have similar types of impact: carp and goldfish increase disturbance of benthic substrates and alter species composition as a result (Wydoski and Whitney 2003). A stickleback species pair in Enos Lake, Vancouver Island, collapsed to a hybrid swarm an event thought to be related to recent introduction of the signal crayfish—modification of benthic habitats by the crayfish is believed to be the primary mechanism of species collapse (National Recovery Team for Stickleback Species Pairs 2005).

3.3 Economic Impacts

The ecological effects of non-native species translate into economic costs through loss of agricultural production, fisheries, forestry, and grazing capacity, plus the impact of human diseases, and many indirect and unquantifiable costs (Mack et al. 2000). Yet, the economic impacts of non-native species are poorly described and quantified (OTA 1993). Economic data on non-native species are heavily weighted toward direct market effects and government control costs. Other costs can be substantial (e.g., North Carolina homeowners spent an estimated $11 million annually to protect residential trees from the European gypsy moth, OTA 1993), but are difficult to obtain, or in some cases impossible to monetize. For example, there is no straightforward way to monetize biodiversity "values," so there is no easy way to include the cost of say, species or population extinctions where the species was not harvested. How also would one objectively monetize human deaths from the introduction of West Nile virus?

The most widely cited estimates of economic costs from non-native species are in OTA (1993) and Pimentel et al. (2000, updated in Pimentel et al. 2004). The OTA study indicates that conservative estimates are at least in the hundreds of millions of dollars annually, and as high as several billion in some years. Pimentel et al. suggest that direct annual costs to the US economy are on the order of $120 billion. The discrepancy between the two studies is in part

due to different assumptions and data sources, but the combined message is nevertheless the same: economic losses from non-native species are high. These costs represent a substantial "tax" on potential wealth (Mack et al. 2000). Both studies indicated that the real cost of non-native species is even higher than estimated, because there is no straightforward way to monetize lost biodiversity "values" through species or population extinctions. Also missing from the estimates of both studies were costs associated with translocating species within the boundaries of the US (i.e., the studies considered non-native species that originated beyond the political boundaries of the US).

The direct cost to fisheries resources in the US was estimated by Pimentel et al. (2004) to be $5.4 billion, despite some benefits from non-native species as part of the $69 billion per year recreational fishing industry. This does not include the enormous cost of controlling other aquatic species such as zebra mussel or Asian clam.

We are aware of no estimates of economic impacts specifically for Canada, or for many other areas of the globe. But estimates that exist for the US indicate that economic costs are large, by virtually any standard. It seems reasonable to assume that many of these same impacts occur in Canada, so economic consequences of species introductions are likely to be very large here also. Proper estimates for Canada would likely be useful as support for changes in policy toward restrictions on the import of some species (Mack et al. 2000).

4. VECTORS OF INTRODUCTION

Non-native species are released into new habitat in a variety of ways and for a number of reasons. Some releases are deliberate, but many are unintentional. Once introduced, many species disperse naturally to additional new habitat. In this section we review the primary mechanisms by which species are introduced, and emphasize main routes of dispersal for non-native freshwater fish in BC.

4.1 Natural Dispersal

Dispersal is a fundamental feature of organisms. The mechanisms of dispersal and the distances traveled vary considerably: seeds or spores may be dispersed huge distances by the wind or by hitch-hiking on animals; other organisms disperse much shorter distances at later life stages in their search for available suitable habitat. The natural distribution of organisms is a function of the physical characteristics of the habitat (e.g., presence of physical barriers such as mountain ranges), the mobility of organisms, and the tolerance (physiological and ecological) of individuals to the receiving habitat. Successful introduction of a species is thus fundamentally dependent on overcoming the natural barriers to dispersal. When a self-sustaining population occurs in new habitat, further spread of the species is dependent on these same three factors, but often at a smaller spatial scale. For example, when introduced to an isolated watershed fish will need to overcome additional barriers such as waterfalls or patches of unsuitable habitat if it is to spread within that watershed.

In freshwater fish, dispersal may occur at one or more life stages. For example, riverine broadcast spawners may release vagile eggs that travel large distances downstream. Dispersal may occur at later life stages as larvae are transported passively in currents, or as mobile

juveniles or adults actively seek available habitat for rearing and spawning. Clearly the ability to disperse naturally (i.e., unaided by humans) will vary considerably among different fish species. Some species, like brook trout, have been able to spread considerable distances within BC after initial introductions, whereas other species like goldfish have remained mostly restricted to their original release sites. The ability of a non-native species to disperse naturally in BC waters will play an important role in the management of that species. It will influence the types of control measures that can be successfully utilized, and likely influence the perceived risk of a newly introduced species.

One feature of natural dispersal bears special mention, and that is the introduction of species from outside the political boundaries of British Columbia. BC is not a remote island, and fish species released in other jurisdictions (e.g., US, Alberta) have dispersed naturally into provincial waterways (Crossman 1991). For example, walleye released into Roosevelt Reservoir have subsequently dispersed upstream into the Columbia River in Canada (Crossman 1991). The management of aquatic resources has always required coordinated efforts, and the management of non-native species is no different in this respect.

4.2 Anglers and Fish Stocking Programs

In British Columbia, especially southern BC, species that are popular in recreational fisheries have been introduced for over 100 years. A number of species (e.g., brook trout, rainbow trout, brown trout, cutthroat trout, steelhead, smallmouth and largemouth bass) have been introduced from outside the political boundaries of BC, or translocated from one watershed to another within BC, in an effort to enhance opportunities for recreational angling. These introductions have occurred through authorized fish stocking programs on behalf of anglers and through unauthorized actions of individual anglers or anglers' associations. This is a common source of introductions for freshwater fish elsewhere in western North America: Schade and Bonar (2005) found that one in four individual fish sampled in the 12 western US states were non-native, and the most common non-natives had been introduced for angling. Introductions and supplementation for at least some of these species continue to occur through both authorized and unauthorized fish stocking efforts.

Within the province, there has been and continues to be a large effort to improve angling opportunities (like elsewhere in North America) through the use of authorized fish stocking programs. These efforts have introduced fish species to many wilderness settings that were naturally fishless, devoid of desirable species, or had insufficient abundance to support a strong recreational fishery (Donald 1987; Pister 2001). For over 120 lakes where undesirable native species occurred, the system was intentionally altered (e.g., eradication of "coarse" fish via use of piscicides) to enhance the probability of successful introduction of sportfish (BC Ministry of Environment fisheries database). Stocking of fish into wilderness lakes and streams has been reduced from historic levels, but there are still large and established programs in BC (e.g., federal salmonid enhancement program, provincial fish stocking programs) to supplement natural fisheries production or create new fisheries through the release of hatchery-reared stock. Although the effort is now reduced, and in many cases native biodiversity values receive greater attention, these programs must accept responsibility for widespread historic alteration of fish communities in the province.

Some species introductions occur unintentionally, through the actions of anglers. For example, the use of baitfish in angling is widespread in some jurisdictions and represents a significant vector for the transfer of fish among watersheds (Litvak and Mandrak 1993; Ludwig and Leitch 1996). Baitfish can be introduced to a waterway through their escape or release during angling activities. Litvak and Mandrak (1993) found that almost half of anglers they polled in their study released unused baitfish alive at the end of their fishing day, and that fishing destinations were often outside the natural range of a baitfish species. Ludwig and Leitch (1996) estimated the probability of interwatershed transfer of fish species (between the Mississippi and Hudson) and suggested this is an occurrence that is likely happening thousands or tens of thousands of times per year. British Columbia prohibits the use of live bait in recreational angling, which effectively limits the commercial distribution of this source of non-native species. It seems likely however, that individual anglers in BC harvest and use as live bait smaller fish species. The current role of anglers in the transfer of fish species among BC watersheds is not known, but is presumably much lower than in jurisdictions with large live bait industries.

In efforts to enhance fisheries in BC, anglers have translocated smaller fish species such as sticklebacks and yellow perch believing them to act as a food source for target fish species like salmonids. This seems the likely route by which some species are currently found in systems beyond their natural distribution. The extent to which this unauthorized introduction of fish is now occurring is not known but for some species (especially yellow perch), it is believed to be significant as new systems containing perch are reported each year.

In addition to translocating baitfish, anglers are responsible for the unauthorized introduction of species that are the intended target of recreational fishing. It is difficult to assess the risk of introduction via this route, but it seems the likely reason for the spread of yellow perch, bass and, most recently, northern pike in BC. Some angler groups are actively promoting the sport fishery values of some non-native fish species in BC, which may also encourage the translocation of these species.

Anglers and recreational boaters may inadvertently spread other species also. Although the use of live holding wells in BC is prohibited, transfers can still occur. For example, Eurasian milfoil is spread to new locations through its transfer on the hulls, propellers, and trailers of recreational boats. This species is known to have significant impacts on the aesthetic and fisheries values of lakes. In addition to the ecological effects that may occur from species introductions, there is also a risk of transfer of fish diseases (e.g., whirling disease) when fish are translocated from one watershed to another, or boats and other equipment are used in several watersheds (Modin 1998).

4.3 Release of Pet, Pond and Live Market Fish

A large number of fish species are imported to BC to satisfy demands for ornamental fish in home aquaria or garden ponds. In addition, live fish markets (e.g., Asian markets in large urban centres) import a number of consumable species. These fish are sometimes released into lakes or ponds when the owners no longer want to keep them, or for certain religious ceremonies. In some cases fish (or other aquatic organisms like snails or vegetation) may escape from rearing or holding facilities. Most ornamental fish are of tropical origin and are incapable of surviving in BC waters, although this is certainly not true of all imported fish.

Goldfish, koi, carp, and other ornamental fish are capable of maintaining self-supporting populations in some BC waters, and this is the likely source of some populations of non-native fish species. The introduction of ornamental and market freshwater fish species is of considerable concern in some warmer jurisdictions (like Florida and Mississippi), but the risks in BC have not been described. The risks may be greatest near larger population centres due to the increased presence of fish and horticulture hobbyists, as well as more diverse ethnic communities and markets. For example, a plecostomus (an herbivorous catfish often sold for home aquaria) was reported in Shawnigan Lake recently (see Table 2). A federal risk assessment was conducted on a number of Asian fish species, most notably northern snakehead and various carp species (B. Cudmore, pers. comm.). Habitat suitability suggested that northern snakehead could survive in most waters in BC. This highly predatory market species has already established populations in a number of US states and poses a huge risk to native fish and other aquatic organisms (Fuller 2006).

4.4 Shipping Contaminants

The introduction of zebra mussel to the Great Lakes represents one of the costliest species introductions in Canada, requiring many millions of dollars in annual efforts to control the effects of the infestation, and wreaking considerable ecological impacts (Environment Canada 2004). The introduction was caused by discharge of ballast water originating in the Baltic Sea (International Joint Commission 2002). Many other aquatic species have been introduced to North America via exchange of ballast water, including fish and invertebrates into estuarine waters (e.g., Matern et al. 2002). International agreements have been adopted to prevent ballast water discharge, but for the most part compliance is voluntary and enforcement is weak (Fay 2002). Many species have been introduced to the Great Lakes and to various estuaries and harbours such as San Francisco Bay and Vancouver harbour, but we came across no records of the invasion of inland BC waters through this dispersal route. It seems unlikely that a species would invade a coastal estuary and then spread inland, however, there has been no formal assessment of this risk.

4.5 Aquaculture

Farming of finfish and invertebrates is an economically important industry in British Columbia. The industry is concentrated in marine coastal waters, but important activities occur in freshwater also. For example, some hatchery-reared salmonids are reared in freshwater pens prior to transfer to the marine environment, and some trout species are farmed entirely in freshwater. There are two primary risks of aquaculture in relation to introduction of non-native species: escape of farmed organisms into the wild and unintentional import of non-target species.

Escaped organisms constitute a source of non-native species when individuals are released or escape from their farming environments and successfully establish in new habitats. Atlantic salmon have escaped from aquaculture facilities and appear to have successfully reproduced in BC (Volpe et al. 2000). This species is not yet widely established in BC rivers, but clearly it is possible for escapes from aquaculture facilities to establish in freshwater systems.

A related risk is the escape of native species that have been domesticated. The release of domesticated fish is a concern due to their ability to compete with and mate successfully with wild stocks. Some have noted that domesticated stocks are essentially cryptic species (i.e., genetically very different though morphologically similar) and constitute threats as "exotic" species (Gross 1998). This concern has been expressed primarily with Atlantic salmon on the east coast and in Europe, but the process may occur wherever there are interactions between wild and domesticated stocks. The release of domestic stock from aquaculture facilities has been expressed explicitly as a conservation concern for some fish species in BC (e.g., white sturgeon, FRWSWG 2005).

Aquaculture may also be a vector for non-native species when non-target species are unintentionally imported along with shipments of farmed organisms. Introduction of Japanese oysters to BC brought several unintended invertebrates and plant species, some of which have become significant pests such as the oyster drill and teredo "worm" introduced in oyster shipments in the early 1900s (Quayle 1969; Carlton 1992). Releases of hatchery-reared salmonids have been implicated in the spread of whirling disease into various geographic areas (e.g., Modin 1998), so there is justification for the concern that aquaculture escapes transmit disease to wild stocks. Diseases are one special class of non-native species. At present, several risks from aquaculture in relation to non-native freshwater fish species in BC exist, but have not been well-quantified.

4.6 Biological Control

Species are sometimes released beyond their native range for the purpose of reducing abundance or eradicating an unwanted species. The benefits and risks associated with this control method are discussed in greater detail in Section 5.4. We know of no fish species that have been introduced to BC for the purpose of controlling other species, though it remains a potential source of species introductions and occurs in adjacent jurisdictions (e.g. in Alberta, grass carp (*Ctenopharyngodon idella*) have been placed in Alberta ditches to control weeds, and mosquito fish (*Gambusia affinis*) have been recommended in ponds to control West Nile virus). Given the highly variable success rate of this control method, some of which have had severe negative consequences, it would be wise to undertake a careful risk assessment prior to using of this technique.

4.7 Water Diversions

Numerous projects have been built in North America to divert water from one watershed to another for reasons of power generation, irrigation and drinking water supply, or navigation. Transfer of water from one basin to another has the potential to release species into new habitats by effectively removing an existing dispersal barrier. There are technologies available to prevent entrainment of organisms into water intakes, but the technologies are seldom 100% effective (Therrien and Bourgeois 2000; Clarkson 2004). Probably the best known impact from non-native species through a water diversion is the introduction of sea lamprey into the Great Lakes. This introduction has had many serious impacts on local fish stocks, and millions of dollars are spent each year trying to control sea lampreys (Environment Canada 2004). Large sums are spent on prevention of non-native species in other projects also (e.g., the 540 km aqueduct known as the Central Arizona Project, Clarkson 2004). Large interbasin transfers

occur in British Columbia, but we found no indication that this has led to the introduction of a freshwater fish species into new habitat in the province. This is nevertheless a potential dispersal route for non-native species, and should be considered during the evaluation of proposed interbasin water diversion projects.

4.8 Barrier Removal

This activity is sometimes undertaken to make new habitats available to native fish species, most notably anadromous salmonids. In BC, barrier removal is not a regular occurrence, but as more salmon stocks become threatened the federal fisheries agency is considering opening up habitat to increase freshwater production. Although issues mainly involve species native to BC, resident species and communities may be threatened by introduced anadromous stocks.

4.9 Shipment of Live Organisms

Rapid shipment of live organisms is now possible via courier to almost all regions of the province. Furthermore, the internet-based pet trade including fish and other aquatic species is very active. Although the shipment of live organisms is regulated, it is likely poorly monitored and enforced, making compliance largely voluntary. Shipments of live organisms may occur from the food industry (e.g., live seafood products), pet shops (e.g., ornamental fish), and the aquaculture and horticulture industries (e.g., aquatic plants and animals). Shipments may originate within the country or outside. The extent of such shipping, and the risks of introducing species via this dispersal route is not known.

5. MANAGEMENT OF NON-NATIVE SPECIES

5.1 Control

Efforts to control non-native species fall into three general categories, prevention, maintenance control, and eradication. A wide variety of techniques have been deployed to control non-native species, with varying success, but most efforts underscore the fact that it is usually very difficult to control a species after it has become established, particularly once it has become widespread (e.g., Meronek et al. 1996). In this section we briefly review the three types of control, and the main methods that have been used. Where possible, our examples come from the literature on control of non-native fish species.

5.1.1 Prevention

We classify as prevention those measures that are used to preclude entry of non-native species to new habitats, as well as use of control measures to eradicate relatively small populations of recently released species to prevent further spread. Given the economic and ecological damage wrought by non-native species, an ounce of prevention is almost certainly worth a pound of the cure. Yet, it has been difficult to implement meaningful measures to prevent the introduction of species to new habitats, and no jurisdiction appears to be successful in preventing the introduction and establishment of non-native species. The reasons for this are several, but in many ways the barriers to effective prevention arise from a clash in values, primarily between political-economic interests and environmental concerns, but also from differences in aesthetic

values. Successful policy (i.e. that can be developed and implemented with a strong base of support) for control of non-native species will likely have to acknowledge and navigate these various values.

One of the primary difficulties with prevention is that it is almost always easier to obtain public funds and justify their expenditure on well-known, existing threats than on difficult-to-quantify, potential threats. There is thus a strong tendency to defer control measures until a non-native species is known to be a pest and is geographically widespread — a tendency toward what many managers refer to as a reactive "fire fighting mode." In the case of non-native species, this may allow certainty regarding the threats from a species, but at that point it is often too late for its eradication. Among the public, there is a general ignorance of the scale of economic and ecological problems caused by non-native species. If economic and ecological consequences of non-native species were rigorously quantified and effectively communicated it may be easier to build a "business case" for implementation of specific prevention measures.

An effective plan to prevent species introductions will have four main elements:

1. Identification of threats,
2. Identification of the most likely routes of introduction,
3. Development and implementation of measures to prevent introduction via those routes, and
4. Development and implementation of a response plan for when the species is found in BC waters.

These are discussed in more detail below.

Identification of Threats.— There has been considerable effort expended by ecologists to predict which species or groups of species are most likely to become established and cause economic and ecological damage. The results of these efforts are discussed in Section 2.3. It would indeed be very useful to know exactly which species or type of species pose the greatest threat to a specific ecosystem so that prevention and control efforts can be directed efficiently. But in general the efforts to classify taxa based on likelihood of successful introduction or likely ecological effects have not provided detailed specific information for managers. At this time there are a few generalizations (e.g., Moyle and Light 1996a, b), but detailed predictive power is severely limited (Kolar and Lodge 2001). Current methods used in risk assessments to screen potentially harmful introductions have a high rate of false positives, meaning that a high number of species are identified as likely to inflict damage (Smith et al. 1999). This may be reasonable from an environmental perspective, but not necessarily from other perspectives (Mack et al. 2000).

Risk assessment screening of freshwater fish species may be somewhat easier for British Columbia freshwaters than for many other ecosystems and jurisdictions. Many species tend to be introduced elsewhere first, especially into the United States, which has a greater number of active vectors due to its large population and economy; its warmer climate also makes it more likely to be the recipient of a greater diversity of species. It is thus possible for managers in BC to monitor activities elsewhere and benefit from the experience of others. Effective communication with managers in other jurisdictions is obviously required for such *de facto* screening to work. Risk assessment screening can be completed at different levels of detail,

from simple lists of possible introductions, to detailed predictive mapping showing potentially suitable habitat for each species and predictions of likely ecological and economic effects.

Identification of Dispersal Routes.— When potentially damaging species have been identified, through some sort of risk-screening procedure, it is necessary to understand how they may be brought into the province. A number of these potential routes are discussed in Section 4. Understanding the routes by which species are introduced is a necessary step in preventing their release into the wild. These dispersal routes should be the target of efforts to prevent movement of unwanted organisms.

Measures to Prevent Dispersal.— Developing and implementing measures to effectively prevent the importation of unwanted species is probably the greatest single challenge facing management of non-native species. There are large and well-organized industries that rely on the importation and distribution of exotic species (e.g., the pet, aquaculture, and horticulture industries), there are internet businesses that will courier live animals to almost any destination, and there are well-organized enthusiasts and that actively seek to import and cultivate or translocate non-native species. There are also political barriers, such as international trade agreements that may restrict the use of control techniques that interfere with economic activity. It is reasonable to expect that some stakeholders will oppose the imposition of regulations and measures that would restrict their ability to import and cultivate non-native species. In developing actions to restrict movements of species, managers are dealing with many types of vectors (individuals, associations, industries) with varying political, economic, and aesthetic perspectives and it will be difficult to deal effectively with such a diversity of groups.

Some of the tools available to combat dispersal of organisms include education, quarantine, regulations, and enforcement. Education is the most important, yet it is difficult, in part because it requires a sustained effort. As noted earlier, there is a general ignorance of the scale of economic and ecological problems caused by non-native species. The scale of the problem and the risks involved in releasing non-native species need to be understood by those that are most likely to translocate species. In many cases, the education process is simply about addressing lack of knowledge by disseminating information, such as the unintended consequences of baitfish release (Ludwig and Leitch 1996). In other cases though, the education process will need to be more than simply a communication about potential impacts of species introductions, because one is often confronting different value systems. For example, fish are often introduced by anglers who believe they are doing the "right thing." It is not that they do not value biodiversity; it is just that they believe the benefits of improved angling opportunities outweigh the risks to biodiversity from introduction of a desired sportfish. This view is not inherently "wrong," but it may severely conflict with management objectives for biodiversity, and changing of such values may occur only over fairly long timescales after considerable sustained effort.

Other available tools are more regulatory in their approach. Laws and regulations can be adopted that prevent importation and translocation of certain organisms. For example, it is unlawful to transport or possess live fish in BC without a permit (federal Fisheries Act 1985, BC Wildlife Act 1996), or to use live bait while angling (British Columbia Sport Fishing Regulations 1996). However, the effectiveness of such statutes and regulations ultimately depend on compliance, and people must be aware of the policies if they are to abide by them. The

continued spread of non-native fish species in BC is an indication that the public is ignorant of such policies, is refusing to adhere to them, or both. When policies are unpopular it may be necessary to back them up with enforcement.

In some jurisdictions, importation of certain organisms requires that strict quarantine procedures be followed. Quarantine procedures may be effective for screening organisms prior to their intentional release into new habitats (e.g., as biological control agents, or for cultivation), but one of the primary dispersal routes for non-native species is unintentional, during the process of economic trade (e.g., ballast water discharge, insects on wood products). It is unlikely that significant restrictions on trade activities, such as the widespread implementation of quarantine of imported materials, would be acceptable to political and economic interests (Mack et al. 2000).

Response Plan.— The final step in developing and implementing measures to prevent introduction and spread of non-native species is deciding what steps resource managers should take when a new population of an unwanted species is found in the wild. Such a plan is somewhat different than the type of plan required to manage widely-established non-native species, but there may often be similarities. In essence, a response plan is aimed at containing the spread of, and where possible eradicating a newly introduced species. The efficacy of a response plan rests first on the detection of newly introduced species, which in itself is a tall task. Whether initial detections are from a dedicated and coordinated monitoring program or a network of amateur observers, there will be the need to detect an introduction early and to verify it, prior to responding to it. Implementing a successful "early warning system" is likely to be difficult, especially given current resources for monitoring of ecosystems, and the large areas of the province that are geographically remote.

When a new non-native species or population is detected some form of response may be required, but it may be difficult to develop detailed *a priori* "recipes" for response actions, because situations will vary considerably among locations, species, and resources that are threatened. Although detailed prescriptions may need to be refined for each situation, it is possible to anticipate a number of questions that will require consideration and to provide some guidance for answers to the questions beforehand. For example, small, contained populations of non-native fish have been eradicated with poisons to prevent further spread, but this is not always an option, especially in populated areas where there are often concerns about human health risks. Some of the questions that should be considered include:

- How noxious is the species?
- What resources (e.g., agricultural crops, biodiversity values) are potentially affected?
- How easily will the species spread from this location to others?
- Where is it found? (e.g., is it in an urban area or a remote wilderness location?)
- How likely is it that the species will be reintroduced if it is successfully eradicated?
- What are the available control methods?
- Which methods are most effective?
- Who are the stakeholders? (e.g., who might be affected by use of the different control methods?)
- Which non-target organisms might be affected by different control methods?
- What resources are available for implementation of different control methods?

- What are the risks associated with different actions? (e.g., no action, eradication, etc.)
- Which are the most "acceptable" methods?
- Are there regulatory issues to consider for implementation of control measures?
- What are the objectives of the response plan? (e.g., eradication at all costs?)

It may not be possible to provide detailed guidance for all potential situations, but it is likely possible to develop guidance for responding to newly introduced species under a range of "generic" situations. The development of a response plan for newly identified systems is one objective currently being considered as part of a policy for the management of non-native fishes in BC.

5.1.2 Eradication

Complete eradication of non-native species is often a preferred option, but in many cases it is not a viable alternative unless the introduction is detected early and sufficient effort applied (Mack et al. 2000). Moody and Mack (1988) found that small "satellite" populations of non-native plants generally expanded at a considerably faster rate than larger conspicuous populations, indicating that eradication of isolated small populations is more effective at controlling spread of a non-native species than treatment of a large conspicuous population. Striking early would have considerable benefits for preventing initial establishment of a species in a new habitat. Unfortunately, the practise is generally to ignore a non-native species until it becomes a bona fide pest (Mack et al. 2000).

Eradication has been successful most often in relatively small areas (e.g., extirpation of feral cats and goats from islands, and salmonids in headwater lakes) or in areas in which it is possible to exert considerable physical control (e.g., reservoirs with deep or complete drawdown capacity). Eradication often requires a sustained effort and use of techniques that may be unacceptable to some members of the public, such as treatment with chemicals. Methods used in eradication and control are discussed further in Section 5.

5.1.3 Maintenance Control

Where complete eradication is not viable, but ongoing damages require some form of action, it may be possible to implement a program to reduce abundance and/or distribution of a non-native species to some acceptable level. This form of control has varying success, often depending on the biology of the organism and the level of effort applied. One of the main drawbacks with this form of control is that it does not offer a permanent solution and requires that control measures be continuously applied. Any let up in the control implies further spread of the species or increased impacts from greater abundance. Nevertheless, this type of control is often the only viable option where complete eradication is not feasible with current resources, technologies and contrasting values. This type of control is widely practised in aquatic systems in BC and elsewhere (e.g., Eurasian milfoil harvest, population reduction of unwanted fish species).

5.2 Physical Methods of Control

Harvest.— High mortality rates imposed by intensive exploitation have led to the extinction or near extinction of many species and populations in a wide-variety of ecosystems (e.g., whales, sea otter, bison, passenger pigeon, Eskimo curlew, sturgeon, Atlantic cod, northern abalone, etc.). There are potential opportunities to control the abundance and distribution of species that are attractive to hunters or fishers through the promotion of directed harvest. This may be achieved via changes to fishing and hunting regulations, such as expansion or elimination of catch quotas, changes to restrictions on harvest timing, and changes to allowable methods of harvest.

Fishing may appear to be highly cost-effective and simple, but there are some serious caveats that should be considered before it is implemented as a control method including, effects on non-target organisms, effectiveness of harvest methods, and effects on the perceived social value of the target organism. These problems are well-illustrated with the example of brook trout, which have been introduced to western North America and have aggressively expanded their range at the expense of native salmonids (Dunham et al. 2002).

Brook trout are a highly-regarded and targeted sportfish in many regions, so the use of fishing regulations to aid its eradication and control seems a tempting approach. Yet this method is not likely to be effective if the primary objective is to increase the relative abundance of native salmonids. Brook trout are often more difficult to catch than native salmonids (e.g., catchability of cutthroat trout and bull trout is ~2.5 x higher than for brook trout; Paul et al. 2003), so where brook trout and native salmonids co-occur the native species will be caught by anglers at a higher per capita rate. Even if regulations stipulate that native species are to be released, there may be additional problems. Hooking mortalities in catch and release fisheries are rarely lower than 10%, so managers should consider whether this additional mortality factor is sustainable for non-target fish. In addition, many anglers have difficulty properly identifying fish species (Schmetterling and Long 1999), which results in higher than expected harvest for non-target species.

The biology of the target organism needs to be carefully considered when assessing whether the harvest method will be effective. Brook trout become reproductively mature at a young age and small size. This life history pattern ensures that many small, mature fish will not be vulnerable to angling, effectively creating a reproductive refuge for the population. Paul et al. (2003) modeled the effects of targeted fishing on cutthroat trout and brook trout in a small Alberta stream. They found that catch-and-release fishing with a 10% incidental hooking mortality could lead to the extirpation of cutthroat trout in 5 years, whereas brook trout could indefinitely sustain harvest with no bag limit. The empirical evidence supports this modeling effort. An organized effort by anglers to selectively harvest brook trout from Quirk Creek, Alberta, showed little effect on the brook trout population during the three years of study (Paul et al. 2003). Experience with introduced smallmouth and largemouth bass populations in Oregon, has had similar results (Temple et al. 1998; Govindarajulu 2005).

The use of bounties, whereby a reward is paid out for each individual killed, is a variant on the idea of using harvest pressure to control a target species. It is common for control of terrestrial vertebrates, but is also used for control of some fish species. For example, a bounty exists for northern pikeminnow (a native species) in the lower Columbia and Snake Rivers in Oregon as

part of an effort to increase survival of endangered salmonids (Beamesderfer 2000). The effects of this bounty program are believed to be partly responsible for an improvement in juvenile salmon survival, but the evidence is incomplete (Beamesderfer 2000; Takata and Ward 2002). We came across no literature suggesting that bounties were effective in the eradication of fish populations, though the tool may at times be effective for population reduction, or as part of a public education program.

Managers should also carefully consider the human social dimension before utilizing fishing and hunting regulations as a tool in control of non-native species. Many of the fish species now introduced in BC were originally released to enhance sport fishing opportunities, and they remain highly-prized throughout their native and their introduced ranges (e.g., bass and brown trout fishing in BC, trout fishing in Australia and New Zealand, etc.). There will likely be opposition to declaring some of these species as pests, and experience has shown that anglers can exert substantial political pressure (e.g., Jackson et al. 2004). But perhaps more importantly, there will also be some risk of inadvertent promotion of a species by increasing its attractiveness for anglers. The use of this method ultimately relies on a species being attractive to anglers. It would be ironic if a non-native species became even more attractive and thereby a target for enhancement by anglers and others. This "by-product" effect has been seriously considered as a threat in the recovery of other fish species. For example, there are worries that promotion of sturgeon aquaculture may increase poaching pressure on wild sturgeon by stimulating demand (Fraser River White Sturgeon Working Group 2005); there are worries that opening of an experimental fishery on introduced crayfish may promote pressure to introduce crayfish to new watersheds (National Recovery Team for Stickleback Species Pairs 2005), and where non-native salmonids have been widely introduced in Australia and New Zealand there is stiff opposition from anglers to trout eradication efforts aimed at enhancing native species (Jackson et al. 2004). In general, these "social" threats have not been well-studied, so it is difficult to quantify their magnitude.

Other Capture Methods.— Traps, nets, and electrofishing have all been used in programs to control or eradicate non-native fish species. These methods tend to be size-selective (e.g., net mesh size, trap opening size, electrofisher voltage and frequency settings, anode dimensions); in most cases more than one method is used at a time and repeat applications are usually required to achieve control or eradication objectives. The effectiveness of the techniques varies depending on the target species, the physical characteristics of the waterbody (size, depth, clarity), and the intensity of the effort (Meronek et al. 1996). Water clarity may influence the capture success of salmonids by gill nets in clear high elevation lakes (Parker et al. 2001). The techniques have been successfully used for eradication in small shallow lakes (Knapp and Matthews 1998; Parker et al. 2001; Vredenberg 2004), but generally are used only for population reduction in larger systems (e.g., Meronek et al. 1996; Rask et al. 2003; Neilson et al. 2004). Parker et al. (2001) state that gill-netting may be a viable method for eradication of salmonids in lakes up to 10 ha in area and 10 m depth; Knapp and Matthews (1998) suggest that it may be effective only in much smaller lakes.

Like most other methods of control, effects on non-target organisms can be a concern. Effects on non-target organisms are most pronounced for taxa that are similar to the control target. For example, control programs that target fish run the risk of netting and trapping native fish species that are meant to be the ultimate recipient of enhancement actions. It can be difficult to

deploy these methods in a way that discriminates among species, though knowledge of life history timing and habitat use can help to favourably bias capture rates, through use of different deployment timing and placement. Incidental by-catch may make these methods inappropriate in cases where the conservation target species is already rare. In situations where by-catch is not an issue, the methods may be quite effective and there may be minimal impacts on other taxa.

Water Management.— In regulated aquatic systems (reservoirs and regulated rivers) there is sometimes opportunity to utilize water management as a tool in the arsenal to enhance native species at the expense of non-native species. In some cases, outright eradication is possible, whereas in other situations there may be only opportunity for favouring success of certain species. The effectiveness of water management as a technique for control of non-native species varies with the degree of water management control, the physical characteristics of the regulated system, and the target species involved. This method obviously is not applicable to unregulated systems.

In very small ponds and reservoirs there may be an opportunity to entirely drain the waterbody to eliminate an unwanted species. After re-filling, native species can be re-introduced using individuals salvaged from the reservoir, or from nearby wild populations. The duration of the drawdown may need to be considerable if the target species is especially pernicious (e.g., bullheads). In reservoirs where deep or complete drawdowns are possible there may be large areas of ponded water that permit survival of fish for long periods of time (e.g., see Tisdale 2000). Additional control methods (e.g., netting, trapping, electrofishing, chemical treatment) may therefore also be required. Larger reservoirs and ponds have seasonal or permanent inflows, and these streams may need to be treated if they are not to provide a seeding population to the reservoir following the drawdown treatment.

In some reservoirs it may be possible to manipulate the timing of drawdowns to differentially impact non-native species. The ability to do this depends on operational flexibility of the reservoir (which will itself be limited by the intended function of the reservoir and the timing of inflows) and the biology of the organisms in the system. Many of the non-native fish species in BC are littoral spawners (e.g., bullheads, bass, perch, sunfish), so there may be some opportunities for manipulating water levels during reproductive timing to negatively impact the reproductive success of these species. Care may be required to avoid impacts to native fish species that are littoral spawners (e.g., kokanee, sculpins, sticklebacks) and to other species like salmonids that may require certain water levels to maintain access to tributary streams.

Littoral nesting species require fairly stable water levels during the reproductive season (Bennett et al. 1985). Water level fluctuations of as little as 1 m have been shown to significantly reduce survival of embryos in littoral spawning kokanee (Modde et al. 1997). Some authors (Reckahn 1986; Cohen and Radomski 1993; Winfield et al. 1998) have observed significant relationships between long-term cycles in water levels and abundance of several fish species, although this is not universally true (e.g., McAfee 1976). The causal mechanisms were not investigated in detail in any of these studies, and therefore remain speculative.

Manipulation of flow regime is possible in regulated river systems, where it is sometimes possible to differentially impact non-native species. For example, Fausch et al. (2001) found a

close match between aspects of flow regime, especially flood timing, and the invasion success of non-native rainbow trout in a wide variety of Holarctic systems. Brown and Moyle (1997) found a similar relationship for brown bullhead and green sunfish in the Eel River in California. Abiotic factors in general were considered by Moyle and Light (1996b) to be the main determinants of successful fish introductions in California. There is currently considerable study regarding the effectiveness of flow regime manipulation in determining the relative abundance of native and non-native species in the Grand Canyon (see Walters et al. 2000). Since maintenance of a natural pattern of abiotic disturbance affects native species success and resistance to invasion, there is some credence to the idea that alterations to flow regimes should be avoided, or natural patterns restored in some systems in order to protect native biodiversity (Moyle et al. 2003).

Construction of Barriers.— The construction of barrier dams is most prevalent in the western US, where it is being used to prevent the spread of brook trout into the headwaters of systems with remnant cutthroat trout and bull trout populations (Novinger and Rahel 2003). The method is employed after brook trout are eradicated from a section of stream or before brook trout have invaded an area (Novinger and Rahel 2003). Adams et al. (2000) note several projects that have created bull trout refugia through construction of barriers to prevent upstream migration of brook trout. The main evidence in support of implementing barriers as control method is the observation that native fish communities in many streams are present only where natural barriers prevent upstream migration of non-native species (e.g., Moyle et al. 2003).

There are several concerns with this approach that should be considered before its implementation. Barriers isolate the target native species as well as the invader, and their use runs some risk that migration of the native species is required to maintain the integrity of that population. Hilderbrand and Kershner (2004) show that substantial lengths of stream are required for self-sustaining populations of cutthroat trout and this is likely true for other species, particularly in oligotrophic systems. Other concerns include the fact that such technology is likely only feasible on relatively small streams, and geomorphic processes of the stream may require a very robust (and therefore expensive) structure. The jumping capabilities of the target species need to be well-understood to ensure that any structure built is indeed a barrier to upstream dispersal (Adams et al. 2000). Studies show that remarkably steep sections of stream are not effective barriers to upstream dispersal of brook trout (Adams et al. 2000). Before implementing isolationist measures there should also be fairly strong evidence that a species is actively displacing native species and not merely taking advantage of declines that are occurring for other reasons. For example, following brook removal and construction of barriers there was no population response in native cutthroat trout in tributaries of the Colorado River, Wyoming (Novinger and Rahel 2003). Evidence of direct population-level effects of a non-native species is usually lacking (Dunham et al. 2002).

Explosives.— Explosive charges have the potential to harm or kill fish even when the charges are relatively small and the fish are a considerable distance from the explosion (Falk and Lawrence 1973; Wright and Hopky 1998). The use of explosives is widely known as an effective method for killing fish, and is forbidden under the federal *Fisheries Act* (R.S. 1985, c. F-14). Biotic impacts from an explosion are dependent on size of the charge, distance from the charge, and the species and size of organism. Fish may be injured or killed by the effects of underwater explosions in several ways, ranging from minor injuries such as loss of scales and minor blood

vessel rupture, to major damage such as tearing of muscle tissue, rupture of internal organs, and disruption of the nervous system. The swim bladder is the organ most sensitive to pressure changes from explosions (Wright and Hopky 1998). The effect of explosions on invertebrates is believed to be negligible. Explosives are somewhat size-selective, in that larger fish are generally more susceptible than smaller individuals.

There are several considerations with the use of explosives to control or eradicate non-native species including regulatory issues, effect on water quality, and indiscriminate effects on other species. Explosives such as dynamite are rich in nitrogen and may introduce nutrients to aquatic systems; the effect on water quality is currently under investigation at the Lawrence Livermore National Laboratory, USA (Govindarajulu 2005). Detonation cords have been used in the Eel River in California to control northern pikeminnow (Govindarajulu 2005). Detonation cords have also been tried in other locations, but their efficacy has not been quantitatively assessed in terms of costs and benefits (Govindarajulu 2005). Explosives were used in an attempt to eradicate bass from Caine Ponds (Nooksack River drainage) in the Lower Mainland in 2001. Conditions were thought to be ideal for this use, yet were nevertheless less than 100% effective due to improper detonation and reduced susceptibility of young-of-year fish to shock waves (J. Roberts, Ministry of Environment, pers. comm.). The effect on non-target organisms needs to be considered carefully, as explosives are indiscriminating in their effects and may kill non-target fish and amphibians.

5.3 Chemical Methods of Control

Of the 40 chemicals that have been used as piscicides, only rotenone and antimycin are believed to be appropriate for use in control of non-native fish species in British Columbia (Govindarajulu 2005). There is a long history of use with these chemicals in BC and elsewhere in North America, especially with rotenone in the eradication of fish prior to fish stocking in lakes (Cailteux et al. 2001). The effectiveness of chemical control depends on a number of factors including, temperature, geophysical and chemical characteristics of the waterbody, and variation in susceptibility among species. Govindarajulu (2005) reviews some of the potential issues associated with the use of chemical piscicides, including the intense opposition that can arise from stakeholders with drinking water concerns. Chemical methods are often highly effective, but they generally require considerable efforts in the realms of regulatory approval and community consultation. Effects of chemical toxicants on non-target organisms can be a concern, especially for other fish species, although some poisons like lampricides are very species-specific (Sorensen and Stacey 2004). Managers should be aware of potential consequences if the chemical toxicant is ineffective for total eradication. For example, chemical toxicants were used to control the invasion of fire ants in the southern US. The toxicants had a greater impact on native species than on the invader, so when applications of toxicants ceased the rate of invasion was enhanced due to the absence of the native ant species (Davidson and Stone 1989, cited in Mack et al. 2000).

One particular case of chemical control shows interesting promise, the use of species-specific pheromones, although this is not yet in a commercial formulation (Sorenson and Stacey 2004). Pheromones have been in use for control of insect pests, and different fish pheromones could potentially be used to divert migratory adults, disrupt migration patterns, enhance capture success of different life stages, and disrupt mating processes. The use of fish pheromones

would likely be used to complement other methods (e.g., trapping and netting), as is currently happening for insects (Sorensen and Stacey 2004). One of the great advantages of this method would be its species-specificity, which may help avoid impacts to native species.

5.4 Biological Control

Some of the problems associated with physical and chemical methods of control have been used to promote the utilization of biological control. This approach is somewhat ironic; it is a planned species introduction, whereby a species is released beyond its native range for the purpose of reducing abundance or eradicating an unwanted species, usually a non-native one. The success of biological control in general, has varied from disastrous (i.e., creation of additional pests), to ineffective, to highly effective (Simberloff and Stiling 1996). Biological control has been sharply criticised for its past failures, its simplistic use of ecological knowledge, and its laisser-faire regulatory environment (Simberloff and Stiling 1996).

The argument for pursuing biological control rests in most part on the observation that some non-native species perform better in their new habitat, where they may be "freed" from the competition, predation, and parasitism experienced in their native habitat (Mack et al. 2000; Keane and Crawley 2002; Colautti et al. 2004). Given the general observation that introduced competitors tend to be incorporated into communities with little ecological consequences, whereas predators generally have a greater effect (Moyle and Light 1996a, b), there may be greater opportunity for the use of predatory fishes as a biological control agent in aquatic systems. Unfortunately, in any system it is difficult to predict the outcome of an introduction and many previous biological control agents have had negative consequences for other native species, including extinction of several species (Simberloff and Stiling 1996). Biological control has been deployed much more often in terrestrial and agricultural systems than in aquatic systems. A cautionary tale for the use of biological control in aquatic systems is the widespread introduction of mysid shrimps to enhance lake fisheries. Although they were originally introduced to *enhance* fisheries, these shrimp have become a severe bane to production of fish and fish-dependent wildlife (Spencer et al. 1991). This occurred despite the considerable number of studies completed prior to their release.

A special case of biological control is the production and release of sterile individuals. This approach is used to control a number of agricultural pests. The technique involves the production and release of numerous sterile individuals to "swamp" non-sterile individuals. In theory, the effects can be numerous. Sterile individuals may be predators (or cannibals) of the target species; they may be intra- or interspecific competitors; and/or they may cause wasted reproductive effort if the target species attempts to court and reproduce with sterile conspecifics. It seems likely that this technique would be more effective in closed systems like lakes, but experience indicates it may also be effective in open systems. For example, the effects of hatchery supplementation of salmon species has become controversial because evidence suggests that wild populations can be impacted by hatchery stocks that replace rather than supplement wild production (e.g., Hilborn 1999; Hilborn and Eggers 2000; Weber and Fausch 2003). In effect, this control technique may turn this process into an advantage rather than a drawback.

We came across no references to this method being deployed in the wild as a control technique for fish, but sterile salmonids are routinely used in fish stocking programs (e.g., all brook trout and some rainbow trout now being stocked in BC), so the potential for implementation exists. At this point we do not know whether or to what extent the technique would be effective as a method for control of non-native fish species, and what factors would influence success. The use of this technique seems ripe for field trials as part of controlled experimentation. For example, as experimental units one could select several small closed lakes with self-reproducing brook trout populations and release into them sterile brook trout. Potential treatments are number of sterile individuals released, age or size of sterile fish, and number of repeat releases of sterile fish. It may also be possible to use this method in conjunction with other techniques, such as net and trap capture methods, at various stages of a control program. For example, it may be easier to swamp a self-reproducing population after its abundance has been substantially depleted.

There are potential drawbacks and several unknowns with this method. Not all fish species produced for freshwater stocking programs in BC are being cultured in large numbers as sterile individuals. At present some species are much more available as sterile individuals than others (e.g., brook trout and rainbow trout). There is some evidence that the method used to produce sterile individuals is not 100% successful, so that some non-sterile individuals may be present in each "batch" of sterile fish. For the provincial brook trout stocking program, this has been addressed in recent years by stocking only all-female triploid fish. However, it is unlikely that this technique would be a quick fix for control; rather, it may require several years for competitive displacement to occur, repeated applications may be needed, particularly in larger or open systems, and even where successful it may take several more years for sterile individuals to die off. However, the success of the technique in other situations (e.g., insects) suggests it would be worth investigating its potential in aquatic systems in BC.

5.5 Determinants of Success of Control Efforts

Meronek et al. (1996) reviewed results of 250 fish control projects published in the primary literature, and examined them for success in meeting their objectives. In many cases the objectives were as simple as a change in the standing stock of the target species. This is considerably more modest than complete eradication of a population, which is often an objective of control of non-native species. They found that only 43% of projects were judged to be a success. A full 28% of the studies were too poorly monitored to reliably classify whether the project was a success or not. Determinants of success included the species being targeted, the control method used, the objectives set for a study, and the methods used to quantitatively assess success. Govindarajulu (2005) discusses several of the factors that should be considered when implementing a control program, including the consideration of biological, regulatory, and social elements. A broad understanding of the social and political context of control options can be as important to the success of control efforts as understanding of the biological dimensions (Jackson et al. 2004).

5.6 Summary

A broad range of control options are, in theory, available to control invasive species. In reality, complete eradication is very difficult and generally attainable in only a few systems. Success

will depend on public support, the ability to respond quickly to new introductions, as well the specific biotic and abiotic factors involved. Most eradication and control options are costly and labour-intensive; the most effective approach is therefore to develop a comprehensive prevention strategy.

6. NON-NATIVE FRESHWATER FISH IN BRITISH COLUMBIA

A wide variety of freshwater fish species has been introduced into British Columbia's lakes and streams. Some of these species originate from other continents (e.g., carp, brown trout), some from other regions in North America (e.g., smallmouth bass, yellow perch, brook trout), and still others have been translocated within BC (e.g., rainbow trout, cutthroat trout). Many species have been introduced as part of authorized fish stocking programs for the purpose of enhancing sport fishing, and these programs continue to create or supplement sport fisheries in many areas of the province. Unauthorized introductions have also been responsible for many introductions of freshwater fish, and continue to be a primary factor in the spread of some species. Unauthorized introductions often occur with the intent of enhancing sport fishing opportunities, but also occur through the release or escape of fish from aquaria, ponds, or aquaculture facilities. Table 1 lists the primary freshwater fish species introduced to British Columbia, and summarizes their distribution and likely mode of introduction.

This section of the report provides a general summary of the main freshwater fish species introduced to British Columbia, a description of the databases used to generate inventory data for non-native freshwater fish species in BC, and individual accounts for each of the species. Province-wide trends are discussed briefly in Section 7. The individual accounts found in the later sections (Sections 6.3 to 0) present basic biological information on a species, trends in distribution, mode of introduction, threats to biodiversity, and potential management actions for control of distribution and abundance. The purpose of the individual accounts is to support plans for the management of non-native freshwater fish species in BC. The accounts are presented in alphabetical order of common name and concentrate on species introduced from outside the political boundary of British Columbia. There are several species that have been translocated within BC, some of these widely, such as rainbow trout. These are discussed briefly in Section 6.2, but were considered outside the scope of this project. Finally, single observations of a limited number of species have been documented recently (Table 2). At least two of these species (bluegill sunfish and yellow bullhead) are found in Washington (including immediately south of the border), Oregon and California in numerous naturalized populations (Wydoski and Whitney 2003). While the status of these two species in British Columbia is unclear, it seems reasonable to assume they can become established in this province as well.

Table 1. The primary freshwater fish species introduced to British Columbia. Note: this table lists only those species not native to BC, or not native to large areas of BC. Species translocated within BC are not listed.

Common Name	Scientific Name	Natural Distribution in BC	Non-Native Distribution in BC	Date of First Observation	Original Mechanisms of Introduction
American Shad	*Alosa sapidissima*	None	Region 2 mainly in Fraser River, other spotty locations	1950	Natural dispersal from US waters
Atlantic Salmon	*Salmo salar*	None	Regions 1 and 2 mainly in coastal streams, also Regions 5 and 6	1987	Aquaculture escapes
Black Bullhead	*Ameiurus melas*	None	Regions 1, 2, 4 and 8	1954	Illegal stocking
Black Crappie	*Poxomis nigromaculatus*	None	Region 2 mainly, spotty locations elsewhere	1950	Illegal stocking
Brook Trout	*Salvelinus fontinalis*	None	All regions but focus is southern and interior areas	1947	Authorized stocking
Brown Bullhead	*Ameiurus nebulosus*	None	Regions 1 and 2 mainly, spotty locations elsewhere	1950	Illegal stocking
Brown Trout	*Salmo trutta*	None	Region 1 mainly, spotty locations elsewhere	1951	Authorized stocking
Carp	*Cyprinus carpio*	None	Mainly southern Regions especially 2, 4 and 8	1938	Illegal stocking

Common Name	Scientific Name	Natural Distribution in BC	Non-Native Distribution in BC	Date of First Observation	Original Mechanisms of Introduction
Fathead Minnow	*Pimephales promelas*	None	Region 1 mainly in Fraser River	1955	Illegal stocking – possibly as bait
Goldfish	*Carassius auratus*	None	Region 2 mainly	1954	Illegal stocking – likely via pet/pond trade
Largemouth Bass	*Micropterus salmoides*	None	Regions 2, 4 and 8	1944	Illegal stocking
Northern Pike	*Esox lucius*	Region 9 – Peace, Liard, Yukon	Region 4 – Haha Lake	2005	Illegal stocking
Pumpkinseed Sunfish	*Lepomis gibbosus*	None	Regions 1, 2, 4, 8	1953	Illegal stocking
Smallmouth Bass	*Micropterus dolomieui*	None	Region 1 mainly, also 2, 3, 4, 5, 8	1944	Illegal stocking
Splake[1]	*Salvelinus trutta* × *S. namaycush* hybrid	None	Region 2, 4 and 9	1977	Authorized stocking
Tench	*Tinca tinca*	None	Regions 4, 6 and 8	1954	Authorized stocking
Walleye	*Sander vitreus*	Region 9 – Peace and Liard	Regions 4 and 8	1981	Illegal stocking/movement from US waterbodies
Yellow Perch	*Perca flavescens*	Region 9 – Swan Lake	Regions 1, 2, 3, 4, 8 and 9	1954	Illegal stocking

[1] This hybrid was included here, but is not described separately in the species accounts.

Table 2. Rare observations of non-native species.

Location	Region	Species	Year reported
Green Lake[1]	1	Pacu (genus and species unknown)	2004
Shawnigan Lake[2]	1	Plecostamus (*Panaque cochliodon*)	1996
Silvermere Lake[3]	2	Yellow bullhead (*Ictalurus natalis*)	2005
Osoyoos Lake[4]	8	Bluegill sunfish (*Lepomis macrochirus*)	2004

[1] Data provided by Gavin Hanke (RMBC)
[2] Collected by Alex Peden, identified by Gavin Hanke
[3] Collected and identified by Gavin Hanke
[4] Collected by D. McPhail

6.1 Data and Analyses

Provincial Databases.— Two databases are the main sources of information used to describe distributional patterns of non-native fish species in BC: the provincial record of fish stocking releases, which we refer to in this document as the "releases database," and the provincial database that documents individual occurrences for all fish species, which we call the "occurrences database."

The provincial government maintains an occurrence information for all fish species, based on observations from biological surveys of freshwater systems in BC. The coverage of this provincial database is extensive both in space and time. However, it is important to understand the limitations of the data for examining trends in the distribution and abundance of non-native species. The database has been constructed by pooling data from observations taken over many years, over many locations, and by many people; data have not been collected systematically with respect to time and geography. The data come from multiple sources (e.g., consulting, academic and government biologists) with varying degrees of expertise and experience. Surveys were conducted for a variety of purposes and hence vary in their precision (e.g., detailed biological surveys that target all species and life stages, and simple presence-absence surveys targeting only one or a few species) and geographic locations (e.g., most surveys are near populated areas or in areas near specific development projects); geographic and species coverage is therefore patchy. Furthermore, not all records in the database are complete (e.g., some records in the database do not have dates or an exact location associated with the record), which further constrains our ability to analyse trends.

Occurrence data used in this project originate from three main provincial sources, including Fisheries Inventory Summary System (FISS), Consolidated Waterbody Surveys (CWS) and non-FISS/CWS specialty databases. Both FISS and CWS include data from other provincial data sources including Resource Analysis Branch (RAB), Fish Habitat Inventory Information Program (FHIIP), Field Data Information System (FDIS), lake surveys (BCLAKES) and Small Lakes Assessment Model (SLAM). Data from all of these survey types were previously submitted to the Fisheries Warehouse for inclusion in provincial databases. Provincial release records were ultimately excluded in this compilation as they are considered independently of observation data. UBC Museum observation data and Fisheries and Oceans Canada Atlantic Salmon Watch Program (ASWP) data were also included in the compilation for this project, in

addition to the above data sources. All observation data (excluding ASWP) were compiled on March 3, 2005 by D. Tesch (Head, Fisheries Business Program, Biodiversity Branch) and are considered up-to-date at the time of compilation. New reports of pike and bass since this compilation was originally completed have also been included. Thus, the dataset does not include records entered since this time, which may include recent observations as well as older records recently entered. The compilation of these different sources of data is considered the most up-to-date and complete source of information regarding the presence of all freshwater fish species in British Columbia.

In terms of quality control, data submitted to the Fisheries Warehouse is assumed to be correct in terms of location and species identification (i.e., the data are as good as the ability of field crews to provide correct information). We have not attempted to filter the data in anyway, and all original data have been included. However, we have attempted some quality control by requesting that each regional biologist review the list of observed non-native species compiled for their region and provide comments. Regional responses varied in ability to provide reviews depending on the knowledge and experience in each region, and time constraints.

The above databases were queried for all records of the 17 species listed in Table 1 (plus splake), as well as mosquito fish (*Gambusia affinis*), which has not yet been documented in BC. Data query included all location information (region, watershed code, gazetted name and coordinates), species information (species, numbers observed), reference information, and collection date. Most records had a minimum of watershed code and coordinate information included, and many had a gazetted name for the waterbody. Where coordinates were not included, the Watershed Atlas (http://srmapps.gov.bc.ca/apps/fidq/) was used to obtain coordinates for the waterbody if possible; this provided a coordinate for the mouth of a system in the case of streams and rivers, and coordinates for the outlets of lakes (the center of the lake if the lake is isolated from an outlet stream). Similarly, where gazetted name was included but no code, watershed code was determined where possible. Where dates were not provided, the original reference source was reviewed and in some cases an approximate date (i.e., year) could be determined.

The provincial release records database is the only comprehensive database available that includes all releases for freshwater fish and anadromous trout species conducted in British Columbia. It dates back to 1901, and at the time of the query (conducted by D. Tesch on March 22, 2005) included all releases up to and including 2004, and a few for 2005. This database was queried to obtain release information (species, location, date released) for mapping purposes, as well as to assist in the determination of mechanism of introduction described in Section 4.

Shapefiles for all of the maps in this document were generated in ARCINFO using the coordinate information for the compiled observation data and release records. Native ranges of all freshwater fish species (including cultured species) that are considered native in at least some areas in BC were based on shapefiles created using expert-reviewed range information at the 3rd order watershed level (provided by D. Tesch, Ministry of Environment). These ranges were overlaid on maps with release record data for cultured species. A small number of points were found to have incorrect location descriptors. In these cases, it was assumed that the coordinates were incorrect and the point was manually moved to the correct location, if possible.

All records in the occurrences and releases databases are associated with a "watershed code," a hierarchal 45 digit number, based on branching order, designed to precisely describe a waterbody's relationship to its watershed in the province of British Columbia. Since it is based on branching order, the watershed code is identical for all lake and river segments within a branch. Newer records in the occurrences and releases databases also have a "waterbody identifier" that allow one to distinguish between lakes and stream segments, but unfortunately older records do not have this data component.

All database queries for this project have been based on watershed codes. For this reason, the number of occurrences in a region may be undercounted in data summaries, if a non-native species exists in both lakes and streams with the same watershed code. For example, a non-native species present in Okanagan, Skaha and Vaseux Lakes, and the Okanagan River, would count as a single observation because the count of occurrences is based on unique watershed codes and all waterbodies share the same code. (They are in the same branching order, the mainstem Okanagan River.) Likewise, a non-native species observed in the headwaters of the Fraser River and also in the mainstem Fraser near Vancouver would be counted only once in our data summaries. This limitation in the database cannot be overcome until waterbody identifiers are attached to each record. Readers should therefore be aware that potential undercounting of occurrences mean that the tabulated distribution estimates and trend summaries presented in this document are conservative. However, the maps include all points and represent the true number of observations. Regardless of limitations, the data summaries provide a reasonable representation of distribution and trends.

Despite several shortcomings in the data, we have analysed occurrence and release data in the hopes of capturing some rough trends in distribution of non-native freshwater fish species in BC. We have analysed the data as is, and have relied on all internal quality assurance procedures to review the data for accuracy and completeness. Specifically, all observation data were distributed to the regional MoE staff to provide comments. Response rate was 100%, but staff were only able to comment on a limited number of observations with which they were familiar.

To capture geographic distribution for a species within the province we summarized occurrence records by MoE region (see table below), by querying the database for occurrences within unique waterbodies in each region. To capture trends, we broke the database into three periods, pre-1950, 1951-1975, and 1976-2005, and queried the total number of unique waterbodies in each period. Selection of these periods was somewhat arbitrary, and was based primarily on availability of data. This analysis often showed a trend toward increasing numbers of waterbodies with non-native species. Although the direction of the trend is likely accurate in most cases, we did not use the data to estimate precise rates of change in a species distribution due to the opportunistic nature of the accumulation of data in the database.

Fish stocking records have been maintained since 1901 and authorized releases of fish are entered into a database that includes the species, life stage, and number of individuals released, the receiving waterbody, date, and source of the stock. The version of the database used for this report includes records from 1901 to 2005. Like analyses of the occurrence data, we analysed these data as is, and have relied on all internal quality assurance procedures to review the data

for accuracy and completeness. It seems likely that some authorized releases have not been entered into the electronic database (e.g., due to loss of records, or poor record keeping), particularly releases during the earliest dates. Furthermore, early records are often incomplete in terms of release date and other information. However, records for the past few decades are considered complete. We summarized fish stocking records by MoE region, after querying the database for unique waterbodies, total number of releases, and the first and last date of release.

Table 3. BC Ministry of Environment Regions.

Region	Name
1	Vancouver Island
2	Lower Mainland
3	Thompson
4	Kootenays
5	Cariboo
6	Skeena
7	Williston
8	Okanagan
9	Peace

Evaluation of Provincewide trends.— We have attempted to evaluate in qualitative terms the likelihood of each species to establish in a new habitat, expand its distribution in BC, and impact local biodiversity (see Section 7). Marchetti et al. (2004a, b) undertook a detailed quantitative analysis of the determinants of success of establishment, spread, and impact of freshwater fish in California and found a number of factors to be important determinants of success — factors indicated as important by a wide variety of previous investigations. Unfortunately, there are insufficient data at present to repeat the analysis for non-native fish in BC. Marchetti et al. (2004a) provide their data in an appendix, but use of the full set of these scores did not produce sensible results for BC, perhaps due to biogeoclimatic differences between California and British Columbia.

Although we could not repeat the complete analysis of Marchetti et al. (2004a, b), we developed a set of subjective scores for invasion and impact of BC's non-native fish fauna. To tally the scores we used four factors (human interest, habitat availability, dispersal ability, and biodiversity threat), and rated each as 1, 2, or 3 (i.e., low, medium, or high); each factor is described briefly below. Scores were assigned based on professional judgement in combination with information from the literature (see below). The scores were rolled up into a single value for each species using the sum of squares of the four individual scores. It should be stressed that these are subjective ratings. Our method is coarse and meant mainly to distinguish among those species that are a large threat to local aquatic biodiversity in BC and those that are not.

Human interest in BC was subjectively scored to reflect the likelihood of release. Species were scored high if they are attractive to anglers or have a high probability of escape from culture facilities. For example, rainbow trout were scored highest because they are continuously stocked in large numbers in many BC lakes, whereas common carp were scored low because

they are of minimal interest, have few source populations, and are unlikely to be translocated by humans.

Habitat availability in BC was subjectively scored to reflect the availability of suitable habitats. The score was based primarily on the availability of habitats in BC that are similar to those in the species' native range. Species' scores were not dependent on whether suitable habitats were occupied, so widely-distributed species were not penalized for occupying available habitats. For example, brook trout were scored high, whereas goldfish were scored low.

Ability to disperse naturally in BC was subjectively scored to reflect whether a species was able to expand its range on its own, or whether it is dependent on human-aided dispersal. For example, brook trout have established in many portions of the province in part because of their ability to disperse naturally. Lake-dwelling species, or species with relatively weak swimming ability were given a relatively low score.

Threat to local biodiversity in BC was scored based on two factors, trophic status as adults and the results of Moyle and Light (1996a, b), who indicated that piscivores have the greatest effect on fish community structure and detritivores the least. Omnivores were given an intermediate score. Trophic status as adults was assigned based on data in Marchetti et al. (2004a).

6.2 Translocated Native Species

Several fish species found within the political boundary of British Columbia have been introduced (authorized and/or unauthorized) to waterbodies beyond their native range in BC. These species include arctic grayling (*Thymallus arcticus*), coastal cutthroat trout (*Oncorhynchus clarki clarki*), kokanee (*Oncorhynchus nerka*), lake trout (*Oncorhynchus namaycush*), lake whitefish (*Coregonus clupeaformis*), northern pike (*Esox lucius*), rainbow trout (*Oncorhynchus mykiss*), walleye (*Sander vitreus*) and yellow perch (*Perca flavescens*). In particular, northern pike and yellow perch have very limited native ranges in BC, being restricted to the far northeast region only (Peace River in Region 9).

One of the primary purposes of this document has been to review the biology and biotic effects of fish species that are not native to waterbodies within the political jurisdiction called "British Columbia." Yet, as we have noted earlier, a species that is found naturally within portions of a political entity like BC, but introduced to habitat beyond its natural range is essentially the same process as release of a species from a different province, country or continent. Both involve introductions to habitat beyond the species' natural range, and in most respects these are merely distinctions of scale of transport that run a continuum from small distances to inter-continental biotic exchange. Some of these translocations involve transportation across political boundaries and others do not.

In our review we have presented the biology and biotic effects of non-native northern pike, walleye and yellow perch, which are discussed in detail in Sections 6.14, 6.18 and 0. Although naturally present in the far northeastern region of British Columbia, we chose to include these species in the review primarily because many of the new introductions of these species are occurring a long distance from their native range, and it appears there is a considerable desire

from some sections of society to introduce these species to new areas. We therefore believed there was a need to provide information to managers and the public about distribution trends and potential consequences of such introductions.

We have chosen not to review some other species in the detailed species accounts, such as arctic grayling, lake trout and lake whitefish, because there have been relatively few introductions and there seems to be little pressure to introduce these species to new habitats. Furthermore, these species are no longer stocked by authorized means in BC. We have avoided discussing still other species, like kokanee, cutthroat and rainbow trout in the detailed species accounts, because the task would require a detailed analysis and critique of past and present hatchery and stocking policies, as well as a detailed review of biological effects of introductions of these species in BC and elsewhere, a task well beyond the scope of this project. Nevertheless, we openly acknowledge that these species have been widely-introduced in BC and have likely had a variety of effects, from relatively benign to large and negative, and in some cases potentially positive effects. The following section summarizes each of these species briefly in terms of native and expanded ranges in British Columbia, as well as possible threats they pose. This section does not include federally-managed or cultured Pacific salmon species.

6.2.1 Coastal Cutthroat Trout (Oncorhynchus clarki clarki)

Distribution and Stocking.— The native range of coastal cutthroat trout includes all coastal watersheds in British Columbia as well as some headwater areas in Regions 1, 2, 3, 5, and 6. Figure 2 and Figure 3 present pre-1976 and post 1976 stocking locations, as well as native range, of coastal cutthroat trout in British Columbia. The most obvious change in stocking over time is the significant reduction in range of stocked systems, particularly those outside the native range. Currently, stocking focuses almost entirely on systems within the native range, with almost all stocking concentrated in on the eastern slope of Vancouver Island in Region 1 and the Vancouver/Sunshine Coast areas of Region 2. Stocking continues for both anadromous and resident life histories, occurring in both sea-run coastal streams and lakes, respectively. Unlike anadromous steelhead programs in British Columbia, neither anadromous or lake stocking programs necessarily use local stocks. In fact, the lake stocking programs rely entirely on the Taylor River stock (believed to contain hybrid rainbow trout × cutthroat trout) from Strathcona Park on Vancouver Island, whereas anadromous programs vary.

Threats.— There is some concern over the continued practice of mixing anadromous broodstock from a variety of capture locations in the lower Fraser watershed to produce offspring for release back into these same streams and sloughs. Similarly, broodstock from the Fulford Harbour anadromous run on Salt Spring Island was used until recently to stock coastal systems in the Victoria area for a number of years. These practices threaten the genetic diversity of local stocks through outbreeding and loss of locally evolved co-adapted gene complexes. A number of studies on coastal cutthroat trout have emphasized that this species, more than most other Pacific salmonid species, is highly structured genetically, and populations appear to be structured primarily at the individual stream level (A. Costello, unpublished data; Williams 2005).

Non-local cutthroat trout have also been stocked into systems that originally were fishless, or contained native cutthroat populations and/or other species. Genetic issues are a concern with

these practices, particularly where native cutthroat trout originally existed. Historic stocking of coastal cutthroat trout may have impacted native westslope cutthroat trout in the southeast region of the province, though this issue has not been reviewed.

Introductions of cutthroat trout can impact other aquatic species, as they are voracious feeders. For example, the unusual threespine stickleback populations found in some lakes on the Queen Charlottes and Vancouver Island have evolved in absence of predators such as the cutthroat trout or in balance with an existing native population. The stickleback phenotypes reflect selective pressures associated with presence or absence of native cutthroat trout. Introduction of cutthroat could result in the loss or significant alteration of these phenotypes.

Potential Management Options.— Future management should consider the fine-scale nature of population structure in cutthroat trout and how this may be affected by authorized stocking practices. Where stocking continues, the use of sterile stocks is strongly recommended, particularly where there is the potential to spread or interact with native species. Anadromous programs should be reviewed, and the use of local populations as broodstock source should be considered where programs continue. Stocking lakes with Taylor stock should be terminated where local wild populations of resident cutthroat trout exist.

6.2.2 Westslope Cutthroat Trout (Oncorhynchus clarki lewisi)

Distribution and Stocking.— Westslope cutthroat trout naturally occur mainly in the Kootenay watershed, as well as spotty areas in the Columbia, Kettle and South Thompson watersheds. Figure 4 and Figure 5 show pre-1976 and post-1975 stocking locations as well as native distribution of this species. Similar to coastal cutthroat trout, the greatest change in stocking practices has been the reduction of stocked locations outside the native range. In particular, stocking in Region 2 has been terminated. All stocking in the Kootenays is currently based on offspring of pure westslope cutthroat trout broodstock taken from a single location, Connor Lake. Only lakes are now stocked but some streams were also stocked up until the late 1980s (e.g., St. Mary River, Bull River, Elk River, Kootenay River).

Threats.— The greatest concern with recent stocking practices is the use of a single broodstock to stock all systems. Stocking is now limited to lakes only, but it is not clear if these lakes are closed systems, or if they contain local westslope cutthroat trout populations. If interactions with local populations is possible, there may be genetic concerns, particularly given the fact that this sub-species of cutthroat trout demonstrates a high degree of genetic structuring at the population level (Rubridge and Taylor 2001), similar to the coastal sub-species. Past stocking used a variety of broodstock sources and stocked many rivers and creeks. Thus, genetic impacts may already have occurred. Furthermore, historic stocking practices released westslope cutthroat trout into systems outside of the species' native range, potentially resulting in hybridization with native rainbow trout and coastal cutthroat trout populations. It should be noted that this species is currently listed by the Committee on the Status of Endangered Species in Canada (COSEWIC) as 'Special Concern' in British Columbia, and 'Threatened' in Alberta, in large part due to past stocking practices.

Potential Management Options.— Future management should consider potential effects from authorized stocking practices on the fine-scale nature of population structure in this species.

Where stocking continues, the use of sterile stocks is strongly recommended particularly where there is the potential to spread or interact with native species. Stocking lakes with the Connor Lake stock should be terminated where local wild populations of cutthroat trout exist.

6.2.3 Rainbow Trout/Steelhead (Oncorhynchus mykiss)

Distribution and Stocking.— The native range of rainbow trout and anadromous steelhead together includes all coastal watersheds in BC, as well as the Columbia, Peace, and lower Kootenay systems, and occurs in all MoE regions. The only areas this species is not native to is the Yukon and Liard rivers in the far north, as well as the upper Kootenay River in the far southeast corner of BC. Stocking now occurs in all of these regions. Stocking range for rainbow trout appeared to have expanded over time, particularly in the northern regions of BC, as well as northern sections of Vancouver Island for steelhead (Figure 6 and Figure 7). It should be noted that most recently (i.e., within past 10 years), steelhead programs have been reduced significantly due to poor success and increasing concerns for wild populations. Anadromous programs have typically followed stricter guidelines; all released smolts are marked, all or most broodstock are wild-origin, and only local broodstock is used. Surplus hatchery fish have been released into lakes in the past, although all stocking now involves only smolt releases into the stream of origin.

Rainbow trout stocking programs differ from those of steelhead. Stocking is focussed on small lakes and uses only a few broodstock sources. Small lakes in every region are stocked, though the number varies significantly among regions. Currently broodstock sources originate from Pennask Lake, Blackwater River and Tzenzaicut Lake. In addition, a small Gerrard trout program exists for the Kootenays, and a domestic strain is maintained in the Lower Mainland (approximately 50 generations removed from its wild – mixed origin) to stock urban lakes in the area. Triploid and triploid/all-female stock has been developed and is becoming more popular as a stocking option in many of the regions. However, reproductively viable fish are also stocked in many lakes. More emphasis has been placed on ensuring that lakes are 'closed' in recent years, and streams are no longer stocked with rainbow trout.

Threats.— There are a range of genetic and non-genetic threats associated with rainbow and steelhead stocking programs. With respect to anadromous programs, concerns include 'swamping' of local wild populations with hatchery fish, as well as domestication (genetic and behaviour) and residualization concerns. It is not clear to what extent past programs have affected wild populations. It is evident that a large number of programs, particularly on the east coast of Vancouver Island, have failed to maintain numbers of fish sufficient for angling purposes but these population crashes have also coincided with poor ocean survival. The jury is still out on the Living Gene Bank program which was designed as a temporary 'boost' to assist recovery of some populations in serious decline. Elsewhere in BC, anadromous programs are associated with both successful fisheries (e.g., Kitimat and Chilliwack) and with populations that have declined drastically (e.g., Seymour).

Threats associated with rainbow trout and lake stocking programs include interbreeding with native populations of rainbow and cutthroat trout, and ecological impacts on other native species via predation and competition. Use of sterile stocks will minimize genetic concerns, but ecological concerns would likely persist.

Potential Management Options.— A number of steelhead hatchery reviews and modelling exercises are currently underway to evaluate risks and benefits associated with BC's steelhead programs. It is hoped that these studies will provide guidance to steelhead management in the future. With respect to lake stocking with rainbow trout, use of all-female stocks will eliminate genetic concerns. This practice is highly recommended for future stocking programs, especially in systems where interbreeding with native rainbow and cutthroat is possible. For the most part, stocking occurs in systems that are believed to be "closed," but some lakes may contain native trout or other fish species. A thorough review should be conducted and stocking with rainbow trout should be eliminated in lakes where native rainbow or cutthroat trout occur.

6.2.4 Kokanee (Oncorhynchus nerka)

Distribution and Stocking.— The native distribution of kokanee and anadromous sockeye salmon is similar to that of steelhead (i.e., all systems with ocean access), plus the upper Kootenay and Columbia systems. Stocking of kokanee in the province has been fairly limited compared to other species and almost entirely within the native range of the species with the exception of the east Kootenay watershed historically and the Peace-Williston reservoir more recently (Figure 8 and Figure 9). Currently, kokanee are stocked mainly in Regions 3, 4 and 5.

Threats.— Threats associated with kokanee stocking are limited to genetic and ecological interactions where native kokanee populations exist, and potential interactions (likely as competitors) with other native species. Kokanee are plankton feeders and require lake habitat for rearing, so they are unlikely to spread much beyond the waterbody in which they are stocked.

Potential Management Options.— Under current stocking practices, no major actions are identified.

6.2.5 Lake Trout (Salvelinus namaycush)

Distribution and Stocking.— The native distribution of lake trout in British Columbia is limited to large, cold lakes mainly in central and northern regions. Lake trout are not found in Regions 1 and 2, and are for the most part naturally absent from Regions 4 and 8. It is generally not found in coastal areas of Regions 5 or 6 either. Lake trout stocking in the province has always been limited (Figure 10), mainly because of the species' specific lake habitat requirements. Many prior stocking efforts failed, including all efforts on Vancouver Island. Lake trout were last stocked in 1990.

Threats.— Lake trout are top predators, so stocking of this species has the potential to impact other species through predation, as well as competition for forage species. The species has also been known to interbreed with bull trout where it has been stocked, and displace lake populations of cutthroat trout. However, these threats are unlikely in British Columbia due to lack of present stocking efforts.

Potential Management Options.— Under current stocking practices, no major activities are identified.

6.2.6 Lake Whitefish (Coregonus clupeaformis)

Distribution and Stocking.— The native distribution of lake whitefish in BC is fairly similar to lake trout in that it is found primarily in interior lakes where cooler temperatures prevail. It is absent from Region 1, most of Region 2 (except the most upstream areas of the Fraser), Region 4 and most of Region 8. In addition, it is absent from coastal systems in Regions 5 and 6 (except around the lower Nass). Almost all efforts at stocking were outside of its native range including Vancouver Island, the Lower Mainland area, and the Okanagan, Lower Kootenay and Columbia systems (Figure 11). It is not clear how successful these stocking efforts were, and no stocking has occurred since 1996.

Threats.— Lake whitefish are mainly benthic feeders, exploiting invertebrates and small fish, although some planktonic species have been observed in the diet (Scott and Crossman 1973). The main threats posed from stocking of this species include competition, and possibly predation of small native fish.

Potential Management Options.— Under current stocking practices, no major actions are identified.

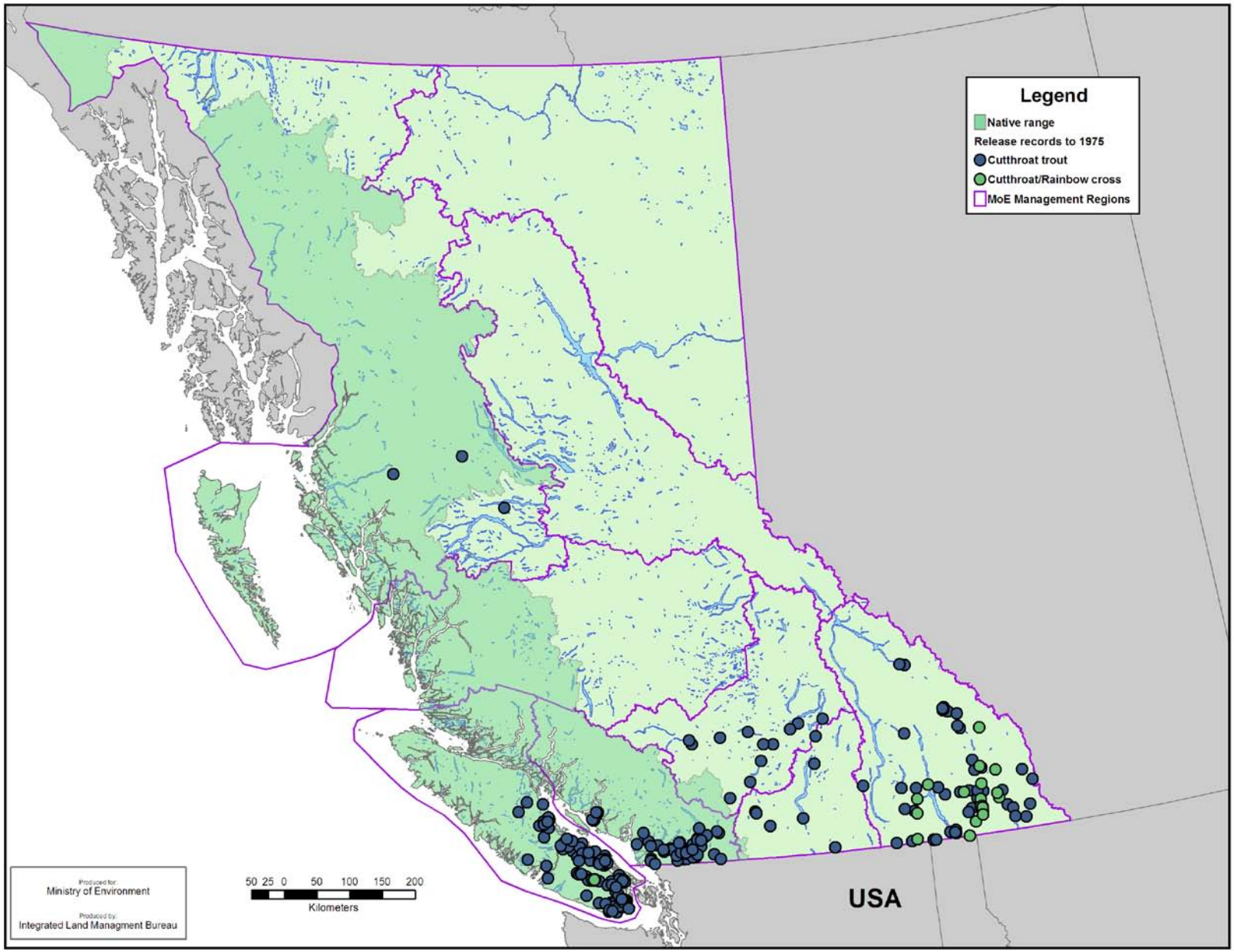


Figure 2. Pre-1976 stocking locations of coastal cutthroat trout in British Columbia, based on the MoE releases database. Also shown is the species' assumed native range.

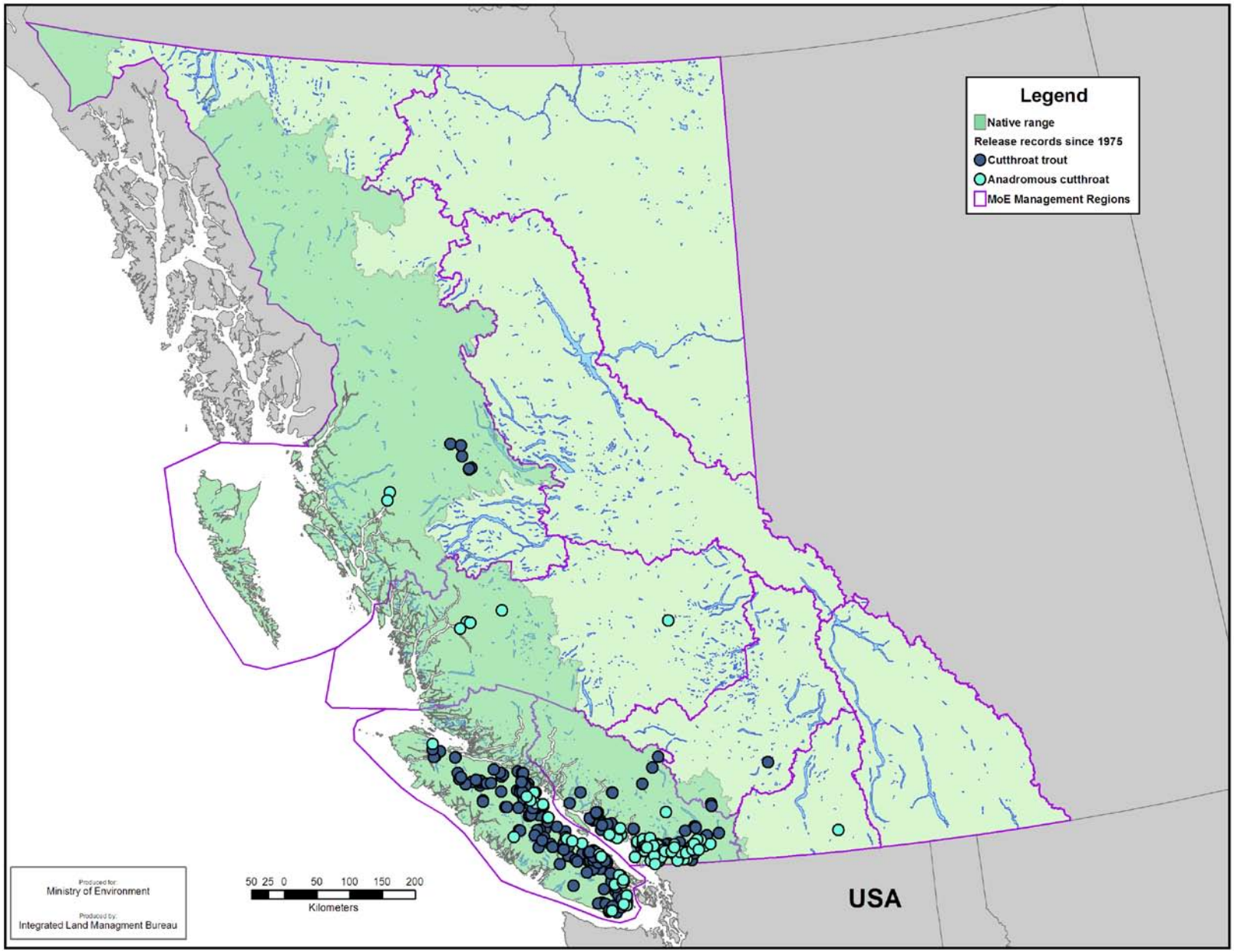


Figure 3. Post-1975 stocking locations of coastal cutthroat trout in British Columbia, based on the MoE releases database. Also shown is the species' assumed native range.

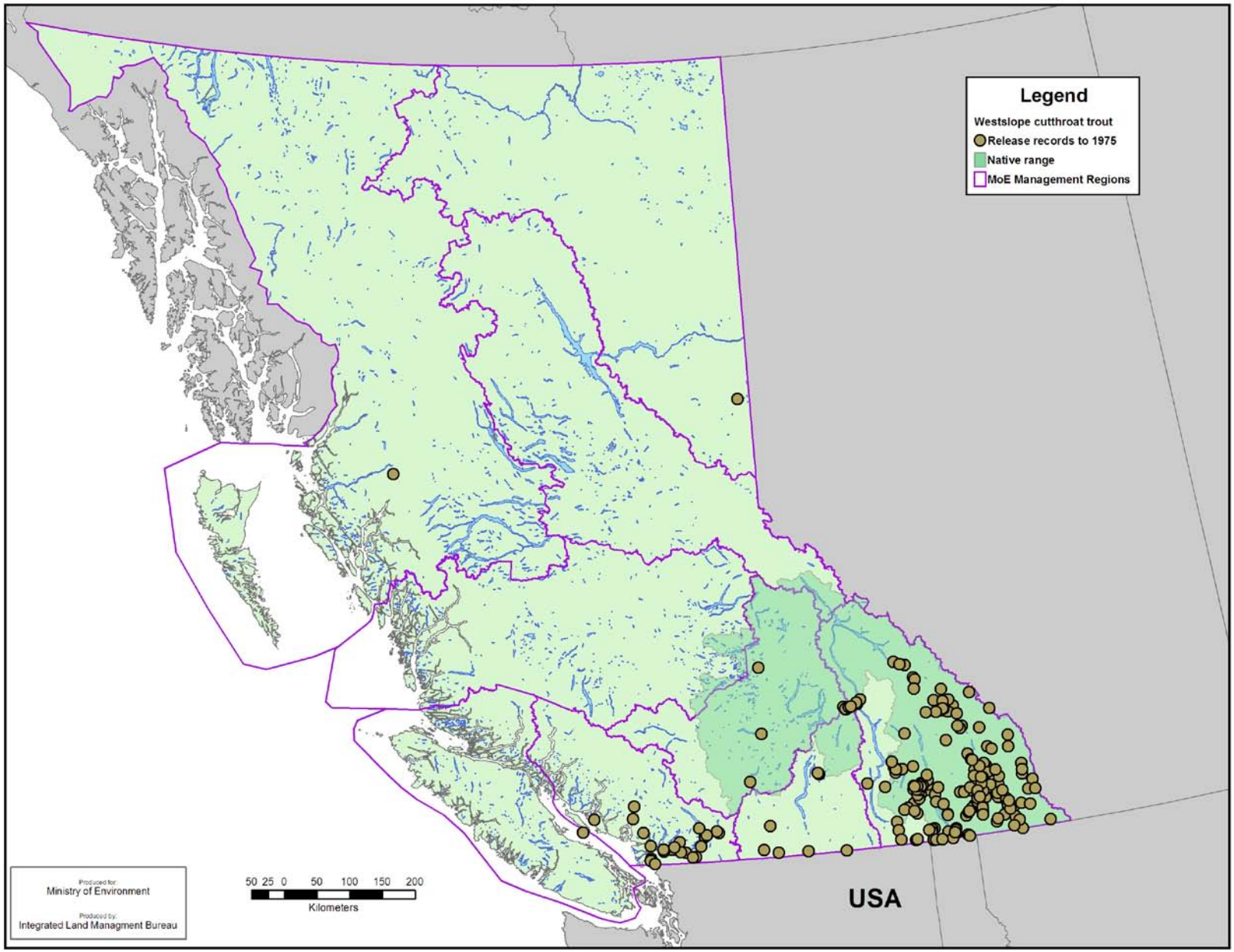


Figure 4. Pre-1976 stocking locations of westslope cutthroat trout in British Columbia, based on the MoE releases database. Also shown is the species' assumed native range.

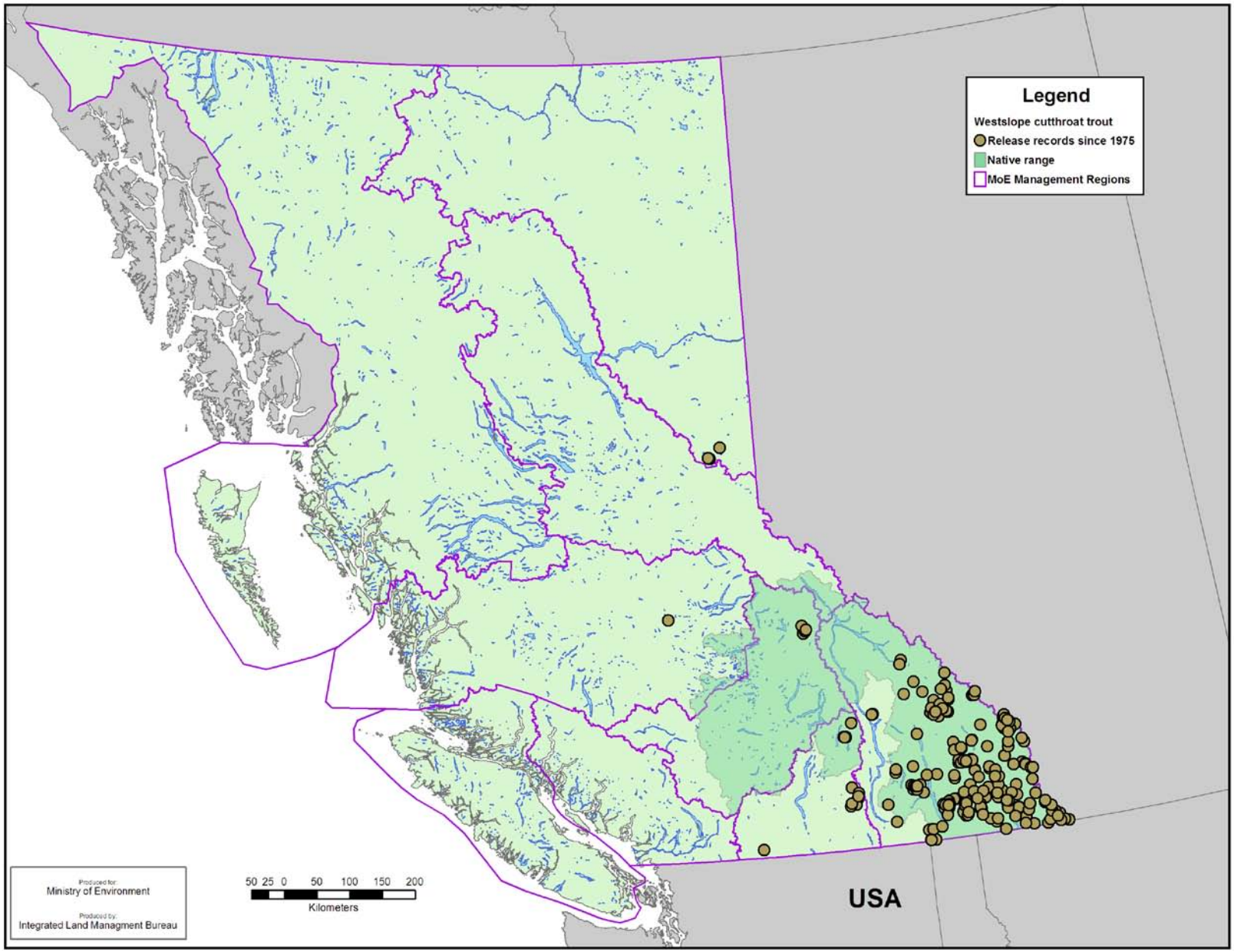


Figure 5. Post-1975 stocking locations of westslope cutthroat trout in British Columbia, based on the MoE releases database. Also shown is the species' assumed native range.

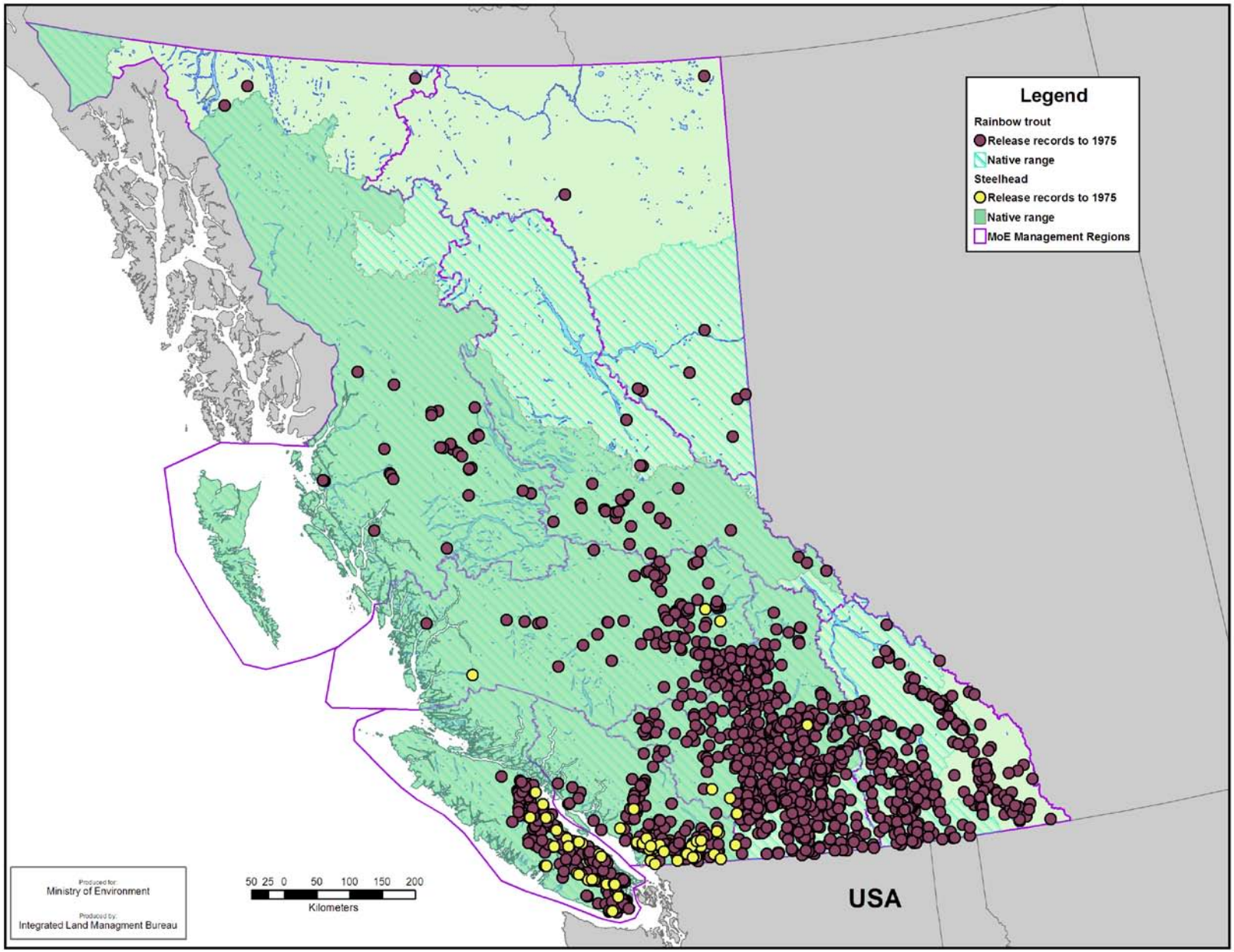


Figure 6. Pre-1976 stocking locations of rainbow and steelhead trout in British Columbia, based on the MoE releases database. Also shown is the species' assumed native range.

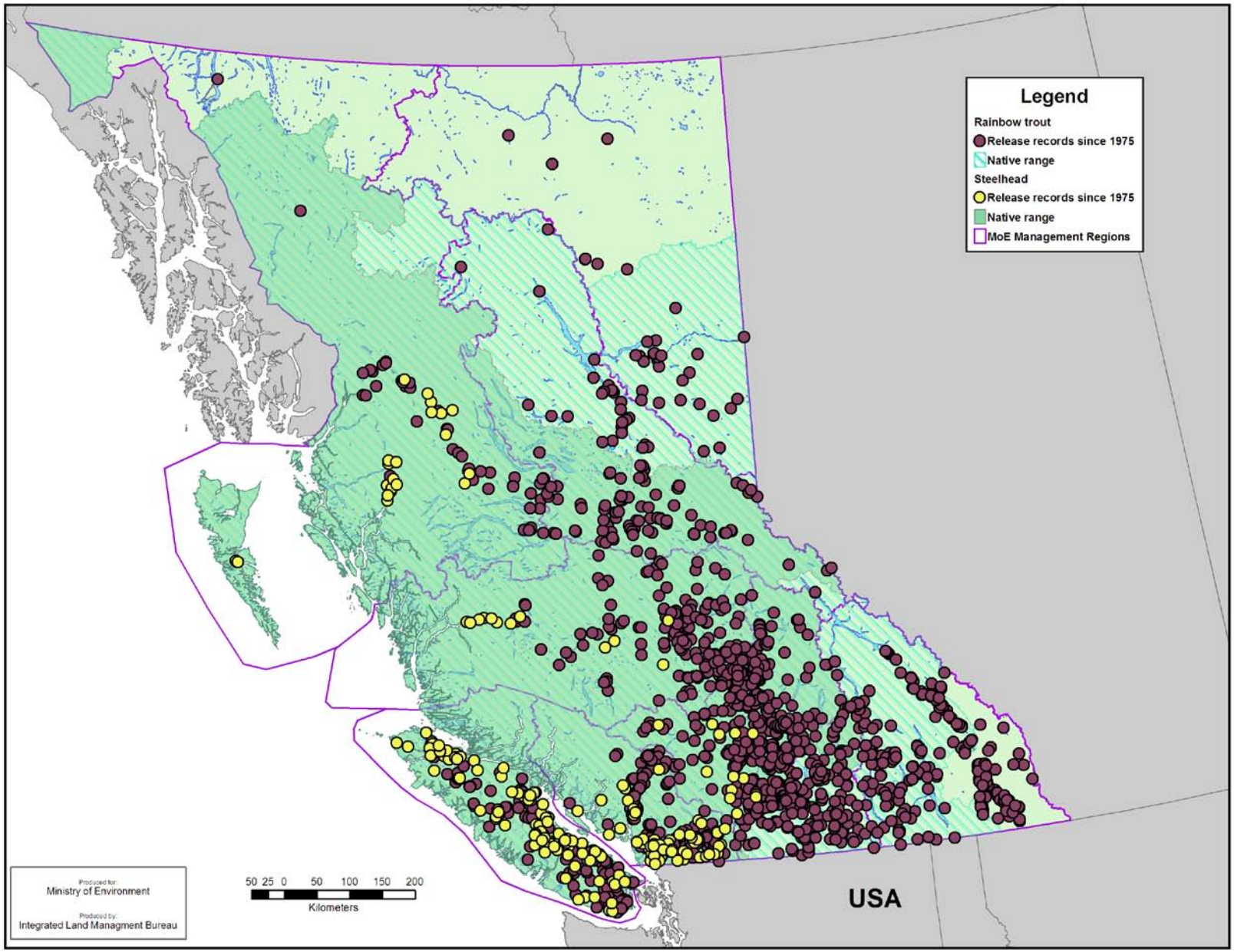


Figure 7. Post-1975 stocking locations of rainbow and steelhead trout in British Columbia, based on the MoE releases database. Also shown is the species' assumed native range.

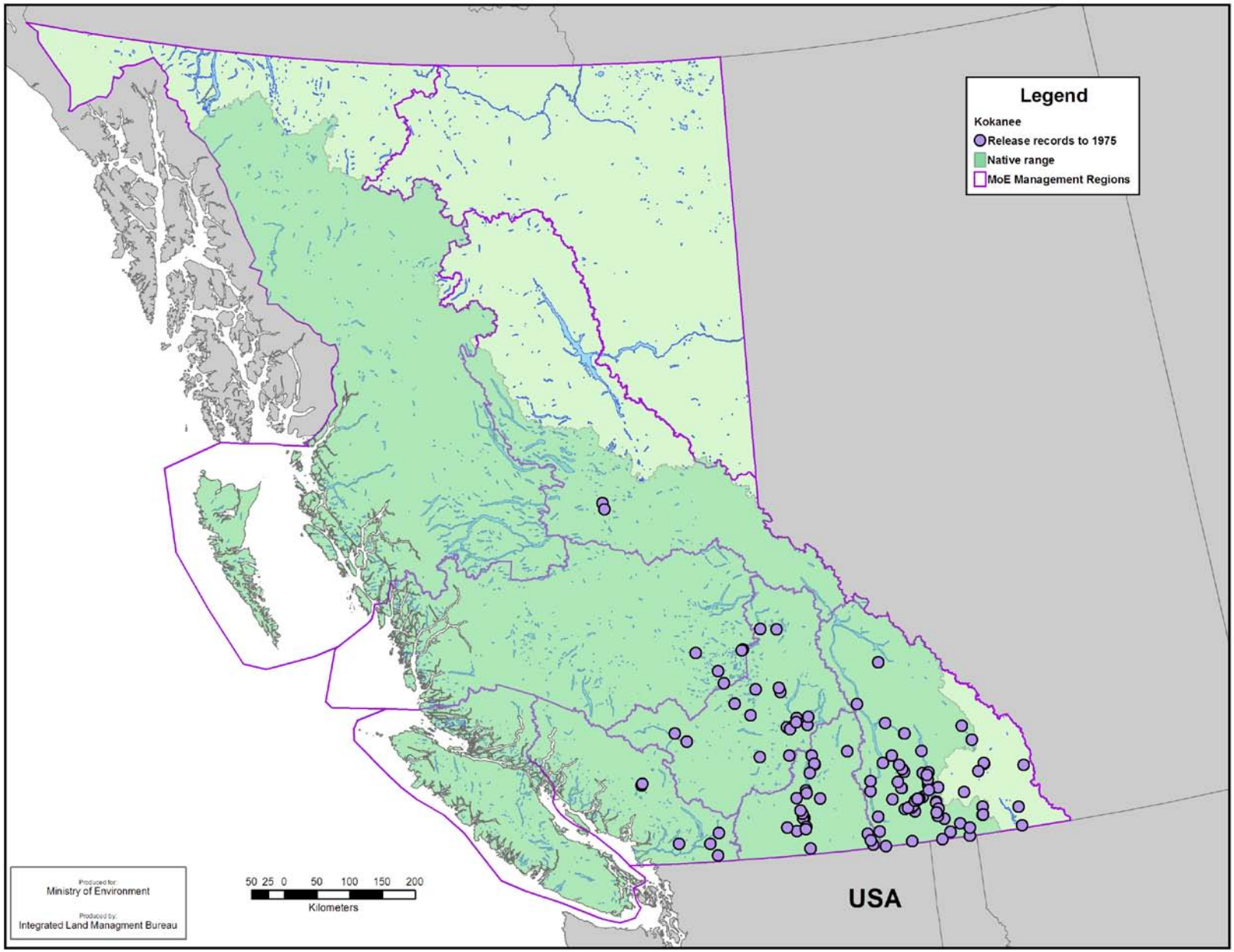


Figure 8. Pre-1976 stocking locations of kokanee in British Columbia, based on the MoE releases database. Also shown is the species' assumed native range.

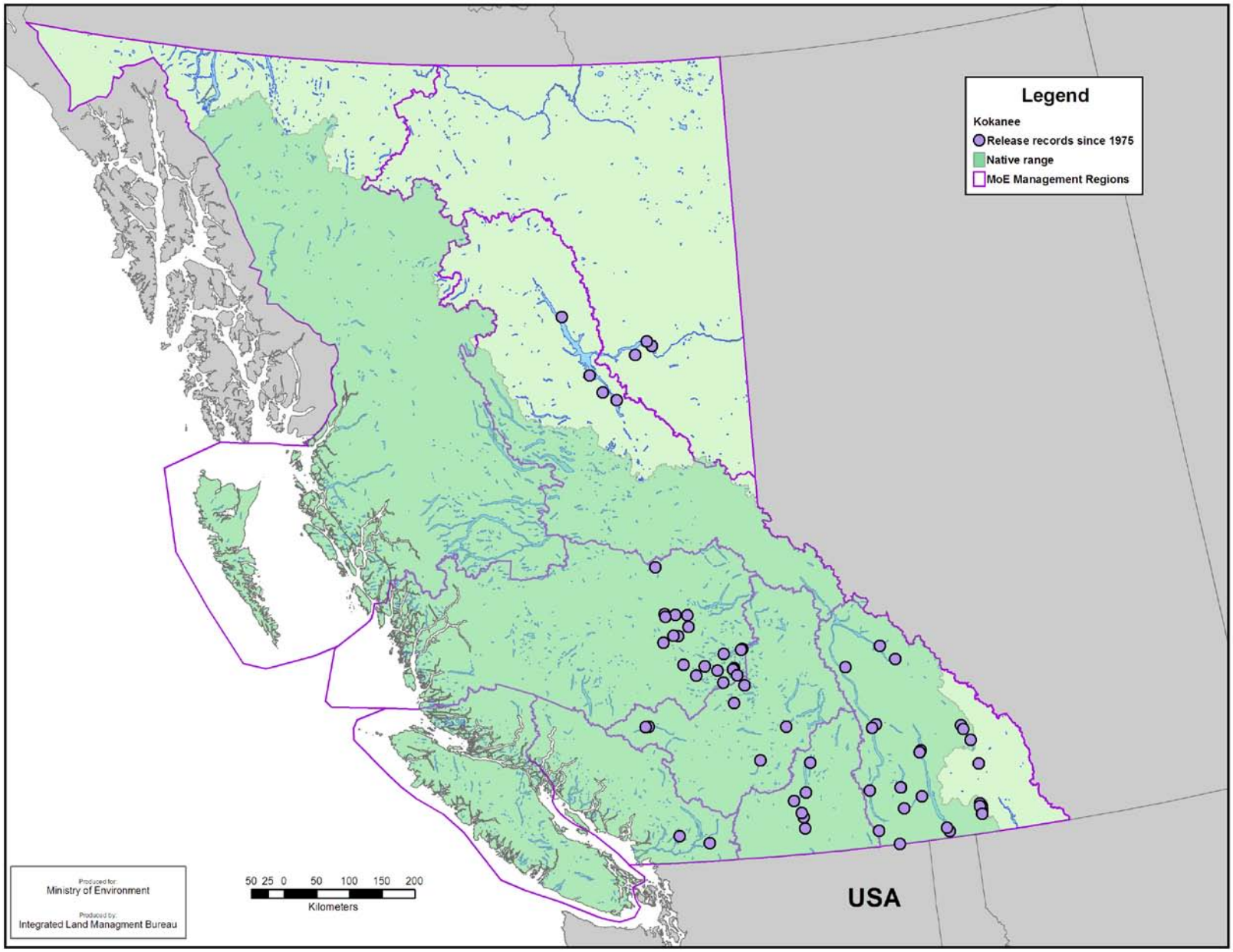


Figure 9. Post-1975 stocking locations of kokanee in British Columbia, based on the MoE releases database. Also shown is the species' assumed native range.

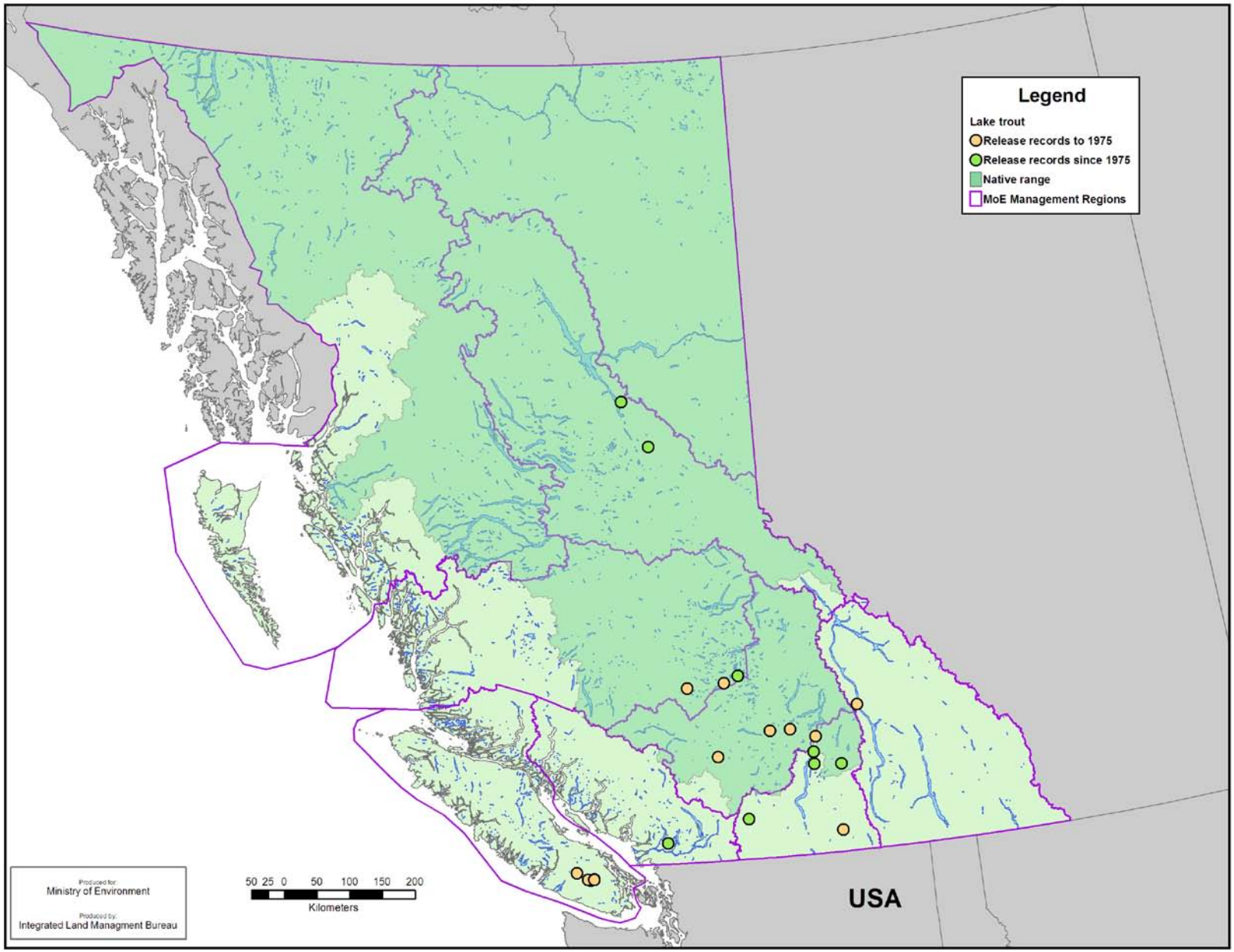


Figure 10. Stocking locations of lake trout in British Columbia, based on the MoE releases database. Also shown is the species' assumed native range.

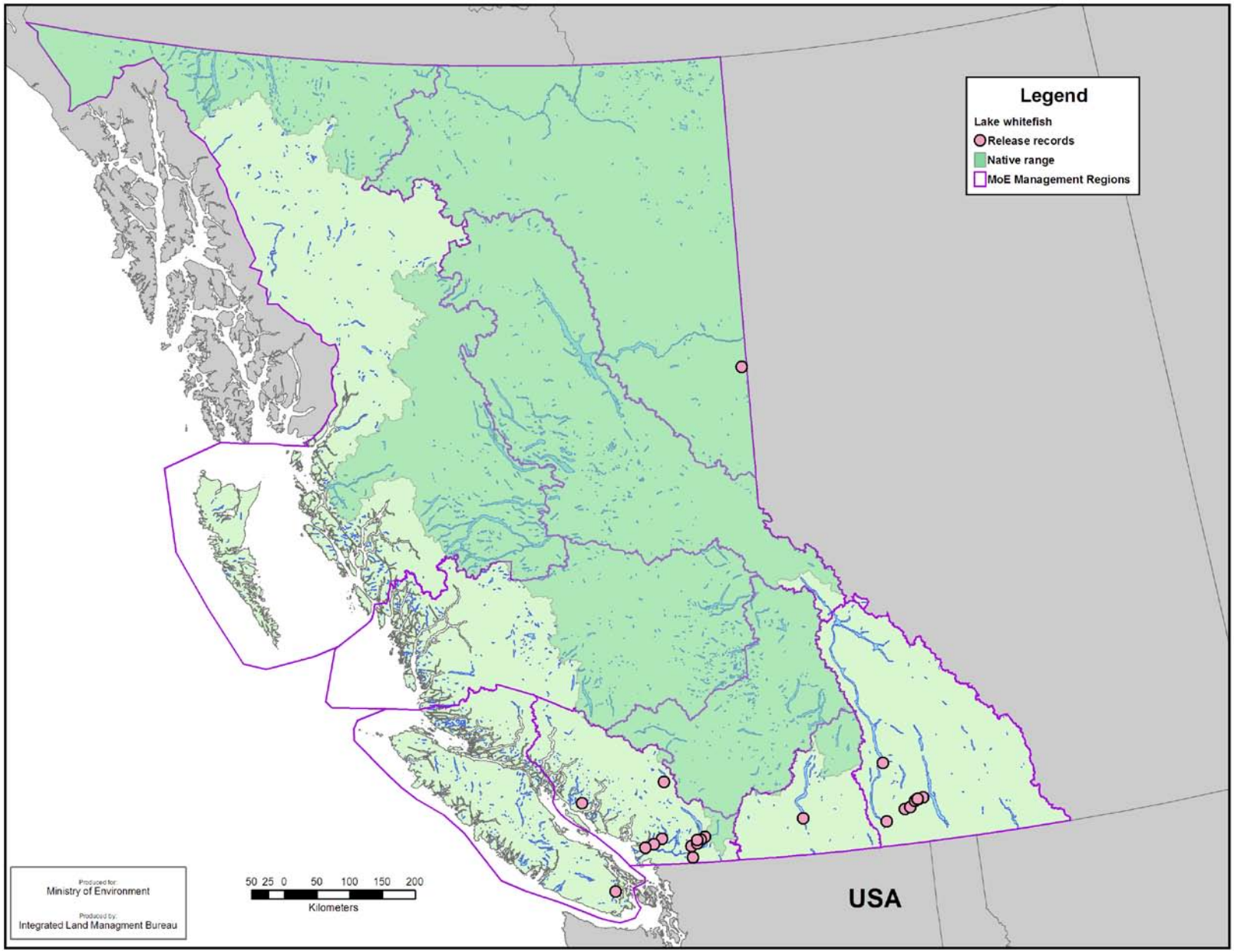
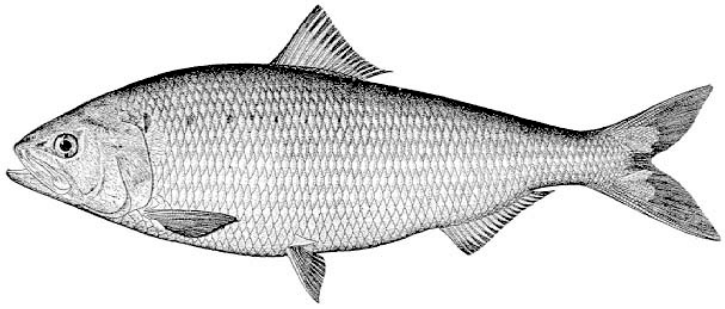


Figure 11. Stocking locations of lake whitefish in British Columbia, based on the MoE releases database. Also shown is the species' assumed native range.

6.3 American Shad (*Alosa sapidissima*)



6.3.1 Taxonomy

Family: Clupeidae (Herrings)
Genus: Alosa
Species: sapidissima
Authority: Wilson, 1811

6.3.2 Habitat

Juvenile Habitat.— American shad are anadromous, and thus begin life in freshwater before migrating to the ocean. Ross et al. (1997) found no relation between relative abundance of juveniles and habitat type, which suggests general use of most riverine habitats. Most measures of physical habitat were not related to juvenile abundance with the exceptions of water temperature in riffles and submerged aquatic vegetation cover (Ross et al. 1997). Temperature is cited (Leggett and Whitney 1972 cited in Stier and Crance 1985) as the primary cue for juvenile emigration from river systems to the ocean. After leaving riverine habitats, juveniles make use of estuarine and coastal habitats, but specific requirements are not well understood, and there is evidence that at least a portion of the population in some systems migrate quickly to offshore ocean areas (Stier and Crance 1985). In developing habitat suitability index (HSI) curves for American shad, Stier and Crance (1985) note that habitat suitability appears to be associated primarily with water quality rather than specific physical habitat features like water velocity, depth and substrate.

Adult Habitat.— Until returning to their natal streams, adult American shad make use of ocean areas only. They form schools and undertake extensive annual migrations in the ocean; distribution is strongly related to water temperatures (Leggett and Whitney 1972 cited in Stier and Crance 1985). They are commonly caught in the trawl fishery, so are assumed to prefer ocean bottom areas (Weiss-Glanz et al. 1986). In the Atlantic Ocean, they occur most frequently in offshore areas of approximately 50-100 m depth (Talbot and Sykes 1958 cited in Stier and Crance 1985; Neves and Depres 1979 cited in Stier and Crance 1985). Little is known about their habits in the Pacific Ocean (Wydoski and Whitney 2003).

Spawning Habitat.— Spawning occurs only in freshwater, but otherwise habitat requirements are very broad (Stier and Crance 1985). Spawning habitat preferences are: 14˚ to 21˚ C, clean sand and gravel, and 0.3 to 0.9 m sec^{-1} velocity. However, HSI curves developed using Delphi methods indicate a very broad tolerance of conditions (Stier and Crance 1985).

6.3.3 Diet

Juvenile Diets.— Juveniles are opportunistic feeders. Ross et al. (1997) noted they appeared to select chironomid larvae and pupae and adult terrestrial insects, and avoid Ephemeroptera. Other studies indicate that they appear to select food more from the water column than from the bottom or surface (Levesque and Reed 1972 cited in Stier and Crance 1985).

Adult Diets.— Adults do not generally feed while in freshwater. In the ocean they are primarily plankton feeders and characteristically feed by skimming — swimming with mouths open and gill covers extended. Their diet consists principally of copepods and mysids supplemented by small quantities of other planktonic crustaceans and some small fishes (Bigelow and Welsh 1925 cited in Stier and Crance 1985; Roy 1969 cited in Stier and Crance 1985; Scott and Crossman 1973).

6.3.4 Life History

Reproduction.— American shad are anadromous, like salmon, and return to their natal stream to spawn (Wydoski and Whitney 2003). However, unlike Pacific salmon, many American shad of both sexes return to the Pacific Ocean soon after spawning (Wydoski and Whitney 2003). They live most of their life and undertake most of their growth in the ocean, and return to their natal stream to spawn in spring or early summer. Time of spawning migrations is correlated with water temperatures, peaking in the Columbia River when water temperatures are around 15.5˚C (Quinn and Adams 1996). American shad are broadcast spawners; eggs are fertilized in the water column and drift downstream (Scott and Crossman; Stier and Crance 1985). Spawning occurs in the evening and at night (Weiss-Glanz et al. 1986). After hatching, American shad spend a portion of their first year in freshwater before migrating to the sea. Some individuals may spend a full year in freshwater, but most leave the river by fall (Weiss-Glanz et al. 1986). Spawning migrations may cover considerable distances as fish will migrate even to the headwaters of some large river systems (Mansueti and Kolb 1953 cited in Stier and Crance 1985). Spawning locations may be as much as 800 km inland, but distribution in east coast rivers is now often limited by dams (Weiss-Glanz et al. 1986).

Age at First Reproduction.— In the Connecticut River, most American shad are 4 or 5 years old when they first spawn, and this is likely accurate for most populations on the east coast (Scott and Crossman 1973; Weiss-Glanz et al. 1986).

Life Span.— The oldest individual recorded was 11 years; the oldest shad in the Bay of Fundy region are 8 or 9 years, and are somewhat older in the Gulf of St. Lawrence (Scott and Crossman 1973). We found no age data for Pacific populations.

Body Size.— American shad are among the largest of the herring family. The largest individual reported was 584 mm; on the east coast individuals are commonly 450 – 480 mm at time of first spawning (Scott and Crossman 1973).

Fecundity.— Females may produce more than 400,000 eggs (Scott and Crossman 1973; Kocik 2000). There is a general cline of decreasing fecundity with distance northward; there is also a trend toward iteroparous reproduction with distance northward (Kocik 2000). All females in southern populations are semelparous (Kocik 2000).

6.3.5 Human Interest

Historically there has been a considerable commercial harvest on the Atlantic coast of North America, but this fishery has declined substantially due to population declines associated with habitat alterations and overfishing (Kocik 2000). American shad are a valued sportfish on the Atlantic coast, and a recreational fishery has been building on the Pacific coast over recent years (PSMFC 1996). Despite the large run size of American shad in the Columbia River, exploitation is not extensive due to protection measures implemented for endangered salmonids (PSMFC 1996).

6.3.6 Distribution

Native Distribution.— American shad are native to the Atlantic coast of North America, where they occur from Newfoundland to Florida. They are most abundant from Connecticut to North Carolina (Weiss-Glanz et al. 1986).

American shad are anadromous; like salmon, they live most of their life in the ocean, and return to their natal stream to spawn. After hatching, American shad spend a portion of their first year in freshwater before migrating to the sea. Spawning migrations may cover considerable distances as fish will migrate even to the headwaters of some large river systems (Mansueti and Kolb 1953 cited in Stier and Crance 1985). Spawning locations may be as much as 800 km inland, but distribution is now often limited by dams (Weiss-Glanz et al. 1986).

Present Distribution.— The American shad still occurs throughout much of its native range, but has been limited by dams and pollution, and there is considerable interest in restoring native habitat for this species (Weiss-Glanz et al. 1986). The American shad was introduced into the Sacramento and Columbia Rivers in 1871, and has since spread to southern California and northward to Cook Inlet, Alaska, and westward to the Kamchatka Peninsula in Asia (Weiss-Glanz et al. 1986). It is especially abundant in the lower Columbia, where recent runs have exceeded 4 million fish (PSMFC 1996).

Most introductions of American shad have not been successful (Scott and Crossman 1973). There is apparently only one "landlocked" population: that introduced to the Millerton Reservoir, of the San Juan River system of California (Lambert et al. 1980 cited in Weiss-Glanz et al. 1986). Introductions into and around the Great Lakes were not successful (Scott and Crossman 1973). Given the substantial success of the introduction to the Sacramento and Columbia, and the subsequent spread of the species, it is clear that other introductions lacked

one or more key ingredients of invasion success such as sufficient numbers (i.e., propagule pressure) or appropriate habitat (e.g., they may not have permitted anadromy).

BC Distribution.— American shad occur with some regularity in the lower Fraser (Scott and Crossman 1973; McPhail and Carveth 1992) and presumably spawn in many coastal streams in BC, though perhaps in low numbers (McPhail and Carveth 1992). There are unconfirmed reports of American shad in the Thompson and Kootenay regions of BC. Figure 12 shows occurrences in British Columbia, based on the MoE occurrences database. McPhail and Carveth (1992) list this species only in coastal rivers.

The following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Region	No. of waterbodies
1	1
2	3
3	1
4	1
5	0
6	1
7	1
8	0
9	0

Distribution Trends.— The following table summarizes time trends based on unique waterbody occurrences in the occurrences database. Note that some records are not dated, so the total number of records is different than the final count in 2005.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
1	1	3	7

Stocking Records for BC.— There are no records of authorized stocking of American shad in British Columbia.

Mode of Invasion.— American shad became established on the Pacific coast of North America through a combination of intentional introductions and subsequent natural dispersal to other drainages. American shad were intentionally introduced to the Sacramento and Columbia rivers in 1871. Individuals dispersed naturally to other drainages and the species now occurs from southern California to Alaska and westward into Kamchatka. It was first caught in BC in 1876 near Vancouver Island (Clemens and Wilby 1961 cited in Scott and Crossman 1973). There are unconfirmed reports of the species in inland waters, but there has been no authorized stocking program for this species anywhere in BC. It seems unlikely that this species is a common target for unauthorized introductions to waterbodies in BC, however, American shad

are a valued sportfish on the east coast and are growing in popularity on the west coast (PSMFC 1996), so it is possible that some anglers may wish to encourage spread of this species.

6.3.7 Threats

Threats from American shad to native fish species have not been quantified or described in any detail, despite their broad distribution and considerable abundance. This in itself may be some evidence that impacts to native biodiversity are not severe. Since there is considerable overlap with native species (including salmonids) in terms of diet and habitat use, it is possible that there are impacts on native biodiversity (Salmon Recovery Science Review Panel 2000), but at this point there is exceedingly little scientific or even anecdotal evidence of negative effects.

6.3.8 Potential Management Actions

There seems little point in developing detailed plans for the control or eradication of American shad until there is some evidence of negative effects from this species on local biodiversity. Some general information in this regard is likely to come from studies that may be planned as part of salmon recovery efforts in the Columbia River in Washington State. Should such studies indicate significant negative impacts from this species, management actions can be prepared in part based on control and eradication options discussed in Section 5. Efforts should be made to prevent unauthorized introduction of American shad into inland waters of BC, but this would likely be best accomplished as part of a more widespread effort to educate anglers and the general public about the potential consequences of translocating species.

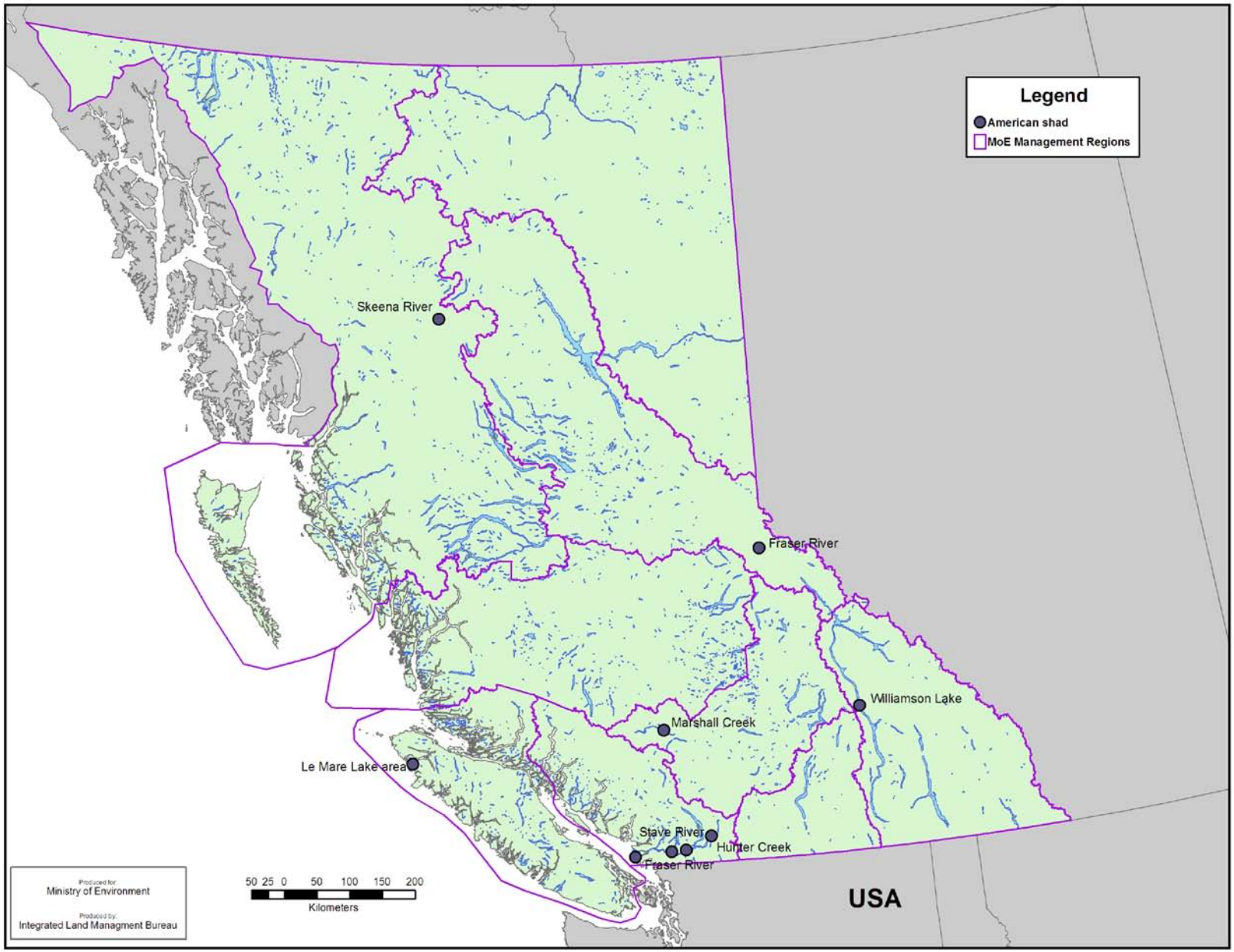


Figure 12. American shad occurrences in British Columbia, based on the MoE occurrences database.

6.4 Atlantic Salmon (*Salmo salar*)



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6.4.1 Taxonomy

Family: Salmonidae (Salmonids)
Genus: Salmo
Species: salar
Authority: Linnaeus, 1758

6.4.2 Habitat

Juvenile Habitat.— In their native range, Atlantic salmon have several life history forms: freshwater-resident, nearshore marine/estuarine-resident, and migratory. All life history forms begin life in cool, clean, well-oxygenated gravel-bottomed streams. Upon emergence from the gravel in the spring (i.e., once the yolk sac has been absorbed), the young remain in their natal stream for a variable period of time before emigrating. For anadromous fish, this ranges from 1 to 8 years depending on water temperature (Jonsson and Jonsson 2004). Prior to outmigration, juveniles undergo smoltification then migrate to the ocean, where they may remain closely associated with the estuarine environment or migrate great distances (>2,000 km). Migratory forms typically inhabit marine areas off the southwest coast of Greenland where they feed for 1 to 4 years prior to returning to natal streams (Jonsson and Jonsson 2004). For freshwater-resident populations, fry typically emigrate from natal tributaries to associated lakes. In this case, they remain in the shallow upper layers of the lake following ice break-up, gradually moving to cooler deeper waters with the onset of summer (Scott and Crossman 1973).

In the stream, there are both seasonal and ontogenetic shifts in use of habitat by juveniles (Maki-Petays et al. 2004). Water velocity at the snout appears to be the principle microhabitat component determining habitat suitability for juveniles (summarized in Armstrong et al. 2003). Smaller fish (4-6 cm) prefer lower velocity water (<20 cm s^{-1}) throughout the year, while larger fish (7-17 cm) prefer faster velocities in the summer, but velocity preferences of large and small fish converge in winter. Similarly, larger fish prefer deeper water than small fish in the summer, but all size classes prefer deeper water in the winter (30-50 cm; Maki-Petays et al. 2004). Temperature tolerances are high: growth of juveniles in freshwater has been observed at temperatures < 1 °C to 17-20 °C (reviewed in Jonsson and Jonsson 2004).

Adult Habitat.— Migratory adults use a variety of migration routes in the ocean to access feeding grounds, either in nearby estuarine areas or at distant marine locations. Freshwater-resident (non-migratory) populations are usually found in land-locked lakes with no access to the ocean. Precociously maturing parr, usually males, remain in the natal stream.

Spawning Habitat.— Females select nesting sites in gravel-bottomed riffle areas above or below a pool. These areas allow them to rest during nest building, which may take several days (Scott and Crossman 1973; Armstrong et al. 2003). Average water velocities for nest sites range from 40 cm s^{-1} to 53.6 cm s^{-1}, and average water depths are 20-50 cm (summarized in Armstrong et al. 2003). Substrate size is variable, depending on water velocities, but nest sites are generally in areas of low fines. Fine sediments are considered detrimental to egg development and embryo survival (summarized in Armstrong et al. 2003). Female Atlantic salmon typically excavate several nests, which combined are called redds, and generally cover an area of 1 -11 m^2 (Armstrong et al. 2003). The eggs remain buried in the gravel to incubate over winter.

6.4.3 Diet

Juvenile Diets.— Young fish in streams generally feed on aquatic insect larvae (especially chironomids, mayflies, caddisflies, blackflies and stoneflies), but terrestrial insects may also contribute to their diet at certain times of the year (Scott and Crossman 1973).

Adult Diets.— The marine diet is much more varied and may include crustaceans (such as euphausiids, amphipods and decapods) and fishes (such as sand lance, smelt, alewives, herring, capelin, small mackerel and small cod). Upon entering freshwater to spawn, feeding ceases (Scott and Crossman 1973).

6.4.4 Life History

Reproduction.— Atlantic salmon are considered iteroparous (i.e., repeat spawners); however most adults spawn only once prior to dying (Jonsson and Jonsson 2004). Spawning occurs in the fall for colder areas (October to December), but may be later (November to March) in less severe conditions, and extend over several months (generally November to March) depending on temperatures (Armstrong et al. 2003). Eggs incubate in the gravel until spring; typically requirements are 450-500 ATUs. After hatching, embryos remain in the gravel until all of the egg yolk is absorbed, finally emerging as free-swimming fry in early summer.

In preparation for spawning, migratory males undergo a physical transformation. The head elongates and a hook-like kype develops on the lower jaw. The female excavates up to several depressions in the gravel in which eggs are deposited. Once the eggs are fertilized, the female covers them with gravel. The eggs are relatively large, 5-7 mm in diameter, and somewhat adhesive for a short period post-extrusion (Scott and Crossman 1973).

Age at First Reproduction.— Age at first reproduction is highly variable and dependent on both life history strategy and growth rate. The life history of Atlantic salmon is highly variable and includes the typically anadromous strategy in which fish hatch and rear for 1-8 years in their natal stream prior to undergoing smoltification and migration to saltwater where they spend 1-

4 years feeding and growing prior to return to their natal stream to spawn. Fish that spend only one year at sea prior to spawning are called grilse. Alternatively, a population may remain resident, such as those in land-locked lakes where individuals hatch in lake tributaries and migrate to the lake to feed and grow prior to returning as mature fish to their natal stream to spawn. Finally, some individuals, typically males, mature precociously as parr, and remain in their natal stream to spawn after 1 or more years.

Relationships between early freshwater growth rate and sea age are complex. It appears that within populations, faster growing fish mature earlier. However, among populations, there appears to be a positive relationship between growth rate at sea and age at maturity (i.e., fish that spend more than one winter at sea generally grow faster than those that spend only one winter at sea prior to spawning; Jonsson and Jonsson 2004).

Life Span.— Few fish live beyond 9 years in eastern Canada, but ages of up to 11 years have been recorded in Quebec (Scott and Crossman 1973). Precociously maturing fish have much shorter life spans; generally few survive beyond first spawning (e.g., 1-4 years).

Body Size.— Body size at maturity is highly-dependent on life history strategy. Grilse (fish that spend 1 year at sea) typically weigh 1.5 to 3 kg. Fish that spend longer at sea typically weigh 2.3-9.1 kg, although weights of up to 35.8 kg have been recorded. Land-locked fish are generally smaller (usually < 1 to 2 kg) than anadromous fish, but weights of up to 20.4 kg have been recorded (Scott and Crossman 1973). In freshwater, 1 year old fish are generally less than 100 mm in length, but lengths are variable for older parr, ranging from 113 to 157 mm at 2 years and 174 to 538 at 3 years depending on life history and location (Scott and Crossman 1973).

Fecundity.— Fecundity varies among populations and is not necessarily directly related to the size of the female. On average, a female will deposit about 1600-1800 eggs per kilogram of body weight (Scott and Crossman 1973). Eggs are relatively large and low in number for the size of the adult, and represent a relatively high-energy reproductive investment per egg (Armstrong et al. 2003).

6.4.5 Human Interest

Historically, Atlantic salmon was considered the ultimate sport fish species by anglers on the east and west sides of the Atlantic Ocean, and reached almost aristocratic status as a food species (i.e., 'king of fish, fish of kings'). Commercial fisheries on both sides of the Atlantic Ocean harvested large quantities of this species from the early to mid-1900s. More recently, wild populations throughout their native range have declined dramatically, in some areas becoming extinct, and recent focus has been on industrial aquaculture. For the past three decades, this species has been one of the most common fish species selected for aquaculture globally. In part, this is due to the availability of domestic strains that have been bred for various attributes (e.g., fast growth, high-density rearing), as well as the species' reputation as the best salmon in the world to eat.

As of January 2005, the BC Ministry of Agriculture and Lands (2005) reported 128 licensed marine salmon farm sites in BC. The majority of these tenures are active annually, with Atlantic salmon representing the majority (76.5% in 2003) of production by weight. Most of these sites

are located off the northeast coast of Vancouver Island in the Broughton Archipelago, as well as on the central to northwest coast of the island. A few sites are located just north of Bella Bella. However, the industry is now in the process of developing a large number of sites around the Queen Charlotte Islands anchored by a large hatchery built in Ocean Falls.

6.4.6 Distribution

Native Distribution.— The native distribution of Atlantic salmon spans both eastern and western Atlantic Ocean coasts in the northern hemisphere, ranging from Portugal to the Arctic Circle on the eastern side, and from southern Greenland to the Connecticut River in USA on the western side (Scott and Crossman 1973).

Present Distribution.— The development of Atlantic salmon aquaculture has greatly expanded the global distribution of Atlantic salmon. Since 1984, this species has been farmed in ocean net-pens in 19 countries around the world, including Canada (east and west coasts), United States and Chile, several European countries, Greenland and Iceland, middle east countries, the Russian Federation and Australia. In terms of established, naturalized populations, evidence of colonization is very limited. One land-locked lake population was established in New Zealand, and the development of a sport fishery specifically for Atlantic salmon in Chile suggests that some runs of Atlantic salmon may now be established there (J. Volpe, pers. comm.). The only other evidence of establishment outside their native range comes from Vancouver Island where at least two year classes of juveniles confirmed (based on otolith and DNA evidence) to be pure wild spawned Atlantic salmon were observed in the Amor de Cosmos River in 2000, as well as in the Tsitika and Adam/Eve rivers in 2000 (J. Volpe, pers. comm.). Within their native range, many of the original wild Atlantic salmon populations have been highly compromised (via disease, habitat loss degradation, genetic introgression) or replaced by escaped farmed Atlantic salmon of unknown origin (Mather et al. 1998). Source populations are not restricted to BC, since large escapes have been reported from ocean net-pen aquaculture facilities in Puget Sound, Washington (Wydoski and Whitney 2003).

BC Distribution.— The following table summarizes unique waterbody occurrences of Atlantic salmon within MoE regional boundaries, based on records in the occurrences database. Figure 13 shows occurrences in British Columbia, based on the MoE occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

On Vancouver Island (Region 1) where the majority of sightings have been documented, 65 watersheds are associated with adult escapee salmon and 11 are associated with juvenile sightings; 4 watersheds had both adults and juveniles. Of these juvenile records, 3 of the streams (Amor de Cosmos, Tsitika, Adam/Eve) are believed to represent instream reproduction, since none of the juveniles had characteristics indicative of hatchery-reared fish (such as otolith traits) no rearing facilities are located nearby and all systems have had reports of adult fish. These juveniles were all observed in 2000, but surveys have been few and sporadic and no further observations of juveniles in these streams have been recorded since this time. The remaining juvenile records are associated with systems on which rearing facilities for Atlantic salmon are located. Elsewhere in BC, the only juvenile record was in Lois Lake where juvenile Atlantic salmon are reared. All adult sightings are believed to have escaped from ocean pens.

Region	No. of waterbodies
1	66
2	12
3	1
4	-
5	3
6	4
7	-
8	-
9	-

Stocking Records for BC.— From 1905 to 1935, there were 158 recorded attempts by the BC government to intentionally introduce Atlantic salmon to 45 BC waterbodies in regions 1, 2, 3, 5 and 6 were made. The focus of efforts was streams and lakes on Vancouver Island, where repeated attempts were made in 30 systems, most notably the Cowichan River. Most releases were either eyed eggs or fry. There has been no authorized stocking of this species in BC for 70 years, and it is assumed that all early introduction efforts failed.

Region	No. of waterbodies	No. of releases	Years
1	30	133	1905-1935
2	7	10	1905-1919
3	3	3	1917-1921
4	-	-	-
5	1	1	1921-1921
6	4	11	1921-1923
7	-	-	-
8	-	-	-
9	-	-	-

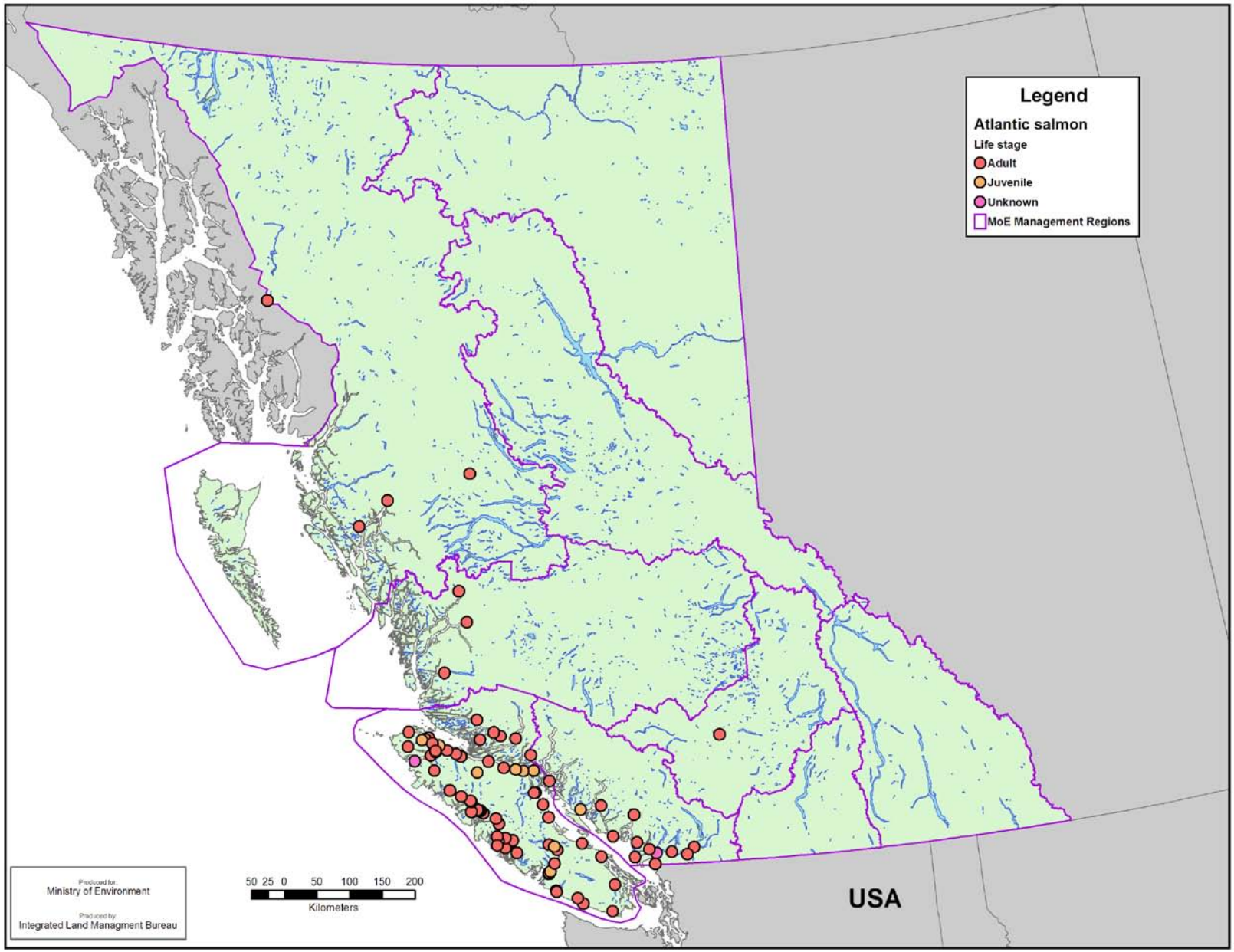


Figure 13. Atlantic salmon occurrences in British Columbia, based on the MoE occurrences database.

Distribution Trends.— The following table summarizes time trends based on unique waterbody occurrences in the occurrences database. Please see Section 6.1 for a description of the database, its limitations, and methods of analysis.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
-	-	86	86

The data indicate a substantial rise in the distribution of Atlantic salmon in BC waters, with virtually all of the observations coming in the last quarter century. Note that the first freshwater observation was for an adult in 1987 and the first juvenile observation was in 1996. The rapid increase in the number of aquaculture facilities since first initiated in 1984 is likely related to the increasing numbers of observations over time, although observation effort also increased over time.

Mode of Invasion.— Although efforts to introduce Atlantic salmon to BC waters in the early 1900s were substantial, it is believed that none of these introductions were successful because no fish were observed subsequently. The failures were believed to be due to a combination of systems being at capacity already with native fish, poor culture techniques and haphazard stocking (J. Volpe, pers. comm.).

The commercial culture of Atlantic salmon in BC's coastal waters was initiated in 1984 (Morton and Volpe 2002). Since this time, Atlantic salmon aquaculture has grown to dominate aquaculture production of salmonids here (other farmed species being coho and chinook). The industry initially imported eggs from established culture facilities in eastern Canada and Scotland, but now maintains its own broodstock in BC for all production. There are a number of hatchery facilities located on Vancouver Island that rear production fish from egg to smolt stage. These juveniles are then transferred to ocean net pens for rearing to market size. Accidental releases occur throughout this process. Hatchery facilities are located on coastal streams, and most juvenile observations are believed to be the result of escapes from these locations. However, the majority of freshwater observations are due to accidental escapes of adult fish, which migrate into anadromous waters when ocean net pens fail. The first documented marine recovery of Atlantic salmon from aquaculture in BC was in 1987, and the Atlantic Salmon Watch Program was established by Fisheries and Oceans Canada and the provincial government in 1991 to record sightings. This program reported close to 400,000 escaped salmon from 1991 to 2001 (Atlantic Salmon Watch Program, Fisheries and Oceans, Canada, 2002), but this estimate is believed to be conservative due to under-reporting (Morton and Volpe 2002). Furthermore, a recent independent survey of commercially caught escapees (i.e., by-catch in salmon fisheries) documented 40.8% more Atlantic salmon than the ASWP in Area 12 fishery in 2000 (Morton and Volpe 2002), suggesting that significantly more escaped salmon occur in coastal waters than are reported by the ASWP.

6.4.7 Threats

Potential threats posed by the presence of Atlantic salmon in coastal freshwater and marine systems include negative ecological and genetic interactions, and disease transfer issues.

Laboratory studies suggest that viability of Atlantic salmon × *Oncorhynchus* species hybrids is limited, but possible (reviewed in Alverson and Ruggerone 1997). The highest viability was for crosses with steelhead females (6.07% survival to hatch); the native west coast salmonid species most likely to share the same habitat type as Atlantic salmon (Alverson and Ruggerone 1997). Regardless of viability, mature Atlantic salmon, particularly males, may still attempt to spawn with native species, and thereby disrupt natural spawning behaviour (J. Volpe, pers. comm.).

Other possible ecological threats include competition, displacement and predation in both freshwater and marine environments. Regardless of their ability to colonize west coast streams, the continued input of large numbers of Atlantic salmon via escapes increases the potential for competition with native species for food and space, in both marine and freshwater habitats. Recent regime shifts in ocean productivity suggest that marine food resources could become more limited in some areas. The combination of limited food resources, reduced native salmon numbers and continued releases (albeit unintentional) of Atlantic salmon may result in greater competition where the artificially maintained numbers of Atlantic salmon reduce resources available to wild species. Colonization of streams could expand this competition to juvenile life stages, particularly for juvenile steelhead. Adult Atlantic salmon are piscivores and could further threaten juvenile migratory forms of native salmonids during marine stages by preying on them.

Currently, the greatest threat posed by Atlantic salmon is the high concentration of two species of sea-lice associated with sea pens. These lice species have always been present in western coastal waters, but the expansion of sea pens in coastal areas has resulted in local population explosions of sea lice. In areas where these pens are located, lice loads on pink salmon smolts may be up to >30,000 times greater than ambient levels (Krkosek et al. 2005) and have frequently been at lethal levels in the past few years (Morton and Volpe 2002). While no data are available, such concentrations of lice could impact other salmonid species such as steelhead, cutthroat trout and Dolly Varden, which remain in coastal areas for extensive periods rather than quickly migrating through. The European ecological analogue of these species is the brown trout which is well documented to have been extirpated in some areas as a result of farm-derived sea lice in areas of intense farm activity (J. Volpe, pers. comm.). No data are available to determine if similar dynamics are occurring in BC with cutthroat trout and Dolly Varden.

6.4.8 Potential Management Actions

The presence of Atlantic salmon in BC coastal waters as a commercially-cultured species has resulted in debates over the extent to which the aquaculture industry threatens native species and ecosystems. The most effective approach to minimize or eliminate all potential threats is to move to land-based aquaculture facilities. Alternatively, the sterilization of all individuals would reduce the potential for colonization or genetic interactions (although some farms would likely still keep broodstock on hand), while the fallowing of farms would reduce problems associated with lice loads. However, these propositions continue to be highly controversial as industry advocates do not believe these threats to be significant. Until there is resolution on the extent of threats to BC's native biodiversity, it is unlikely that significant changes in culture practice will be adopted by the industry.

6.5 Black Bullhead (*Ameiurus melas*)[1]



6.5.1 Taxonomy

Family: Ictaluridae (North American freshwater catfishes)
Genus: Ameiurus
Species: melas
Authority: Rafinesque, 1820

6.5.2 Habitat

Black bullhead is a warm water species found only in freshwater, where it occupies the benthic habitat of lakes, reservoirs, and slow-moving backwaters of low-gradient streams (Scott and Crossman 1973). The black bullhead does not tend to co-occur with other bullhead species (Scott and Crossman 1973). They are even more tolerant than brown bullheads of a wide range of environmental conditions and pollutants (Scott and Crossman 1973). We found no mention in the literature of differential habitat use by juveniles and adults.

6.5.3 Diet

Black bullheads are nocturnal, omnivorous, bottom feeders. They seek food on bottom substrates with the aid of tastebuds on four pairs of barbels surrounding their mouth.

Juvenile Diets.— Juveniles eat insect larvae, crustaceans and leeches (Scott and Crossman 1973).

Adult Diets.— Adults eat insect larvae, molluscs, crustaceans, leeches, plant material and fishes (Scott and Crossman 1973). Black bullhead are considered serious predators of some fishes in the US, including some endangered and game fish (Fuller 2005a).

6.5.4 Life History

Reproduction.— Black bullheads spawn in spring and summer when water temperatures increase to around 21˚ C (Scott and Crossman 1973). Females build nests, and both parents guard and fan the eggs (Scott and Crossman 1973). After hatching, the young are protected by one or both parents for several weeks, until they are about 25mm, after which they are abandoned (Scott and Crossman 1973).

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[1] Image used with permission from Robert Rice, www.nativefish.org

Age at First Reproduction.— We found no information for age of sexual maturity, but given their typically short lifespan, maturity of black bullhead likely occurs at an early age, perhaps one or two years.

Life Span.— Maximum age is listed in Scott and Crossman (1973) as nine years, though less than four years is common. Maximum age is reported in FishBase (2005) as 10 years.

Body Size.— Adult black bullhead tend to be smaller than other bullheads, usually only 130mm to 180mm (Scott and Crossman 1973). The largest black bullhead is listed in Scott and Crossman (1973) as 380mm and approximately 900g. Maximum size is listed in FishBase (2005) as 660mm and 3620g; the reason for such a large discrepancy is not clear.

Fecundity.— Scott and Crossman (1973) list fecundity as 3,000 to 4,000 eggs in the ovaries of a typical female, of which about 200 are laid in the nest at a time.

6.5.5 Human Interest

Black bullheads are of limited interest to anglers in British Columbia, but are an important sportfish in the Midwestern US states (Scott and Crossman 1973). Like other bullheads, they are easily caught with still-fishing methods (Wooding 1994). The species is relatively easy to rear as a pond culture fish (Scott and Crossman 1973). It is considered a nuisance or pest in some locations (FishBase 2005).

6.5.6 Distribution

Native Distribution.— The black bullhead is native to the freshwaters of eastern and central North America. In Canada, the natural range is restricted to Lake Erie and its western tributaries and the Qu'Appelle-Souris-Assiniboine systems in southern Saskatchewan and Manitoba (Scott and Crossman 1973). Elsewhere in North America, the black bullhead occurs naturally in the Great Lakes, Hudson Bay, and Mississippi River basins from New York south to the Gulf of Mexico; and Gulf Slope drainages from Mobile Bay, Georgia and Alabama, to northern Mexico (Fuller 2005a). It is not native to the Atlantic Slope south to Florida (Fuller 2005a).

Present Distribution.— The black bullhead has been extensively and successfully introduced in the western and eastern US, many European countries, Russia and Chile (Scott and Crossman 1973; FishBase 2005). Black bullhead are apparently rare in Washington State (Wydoski and Whitney 2003).

BC Distribution.— The following figure (Figure 14) and table summarize unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1. The database tracks occurrences of black bullhead and brown bullhead separately, but there is also a third category of "catfish – general," for observations that do not

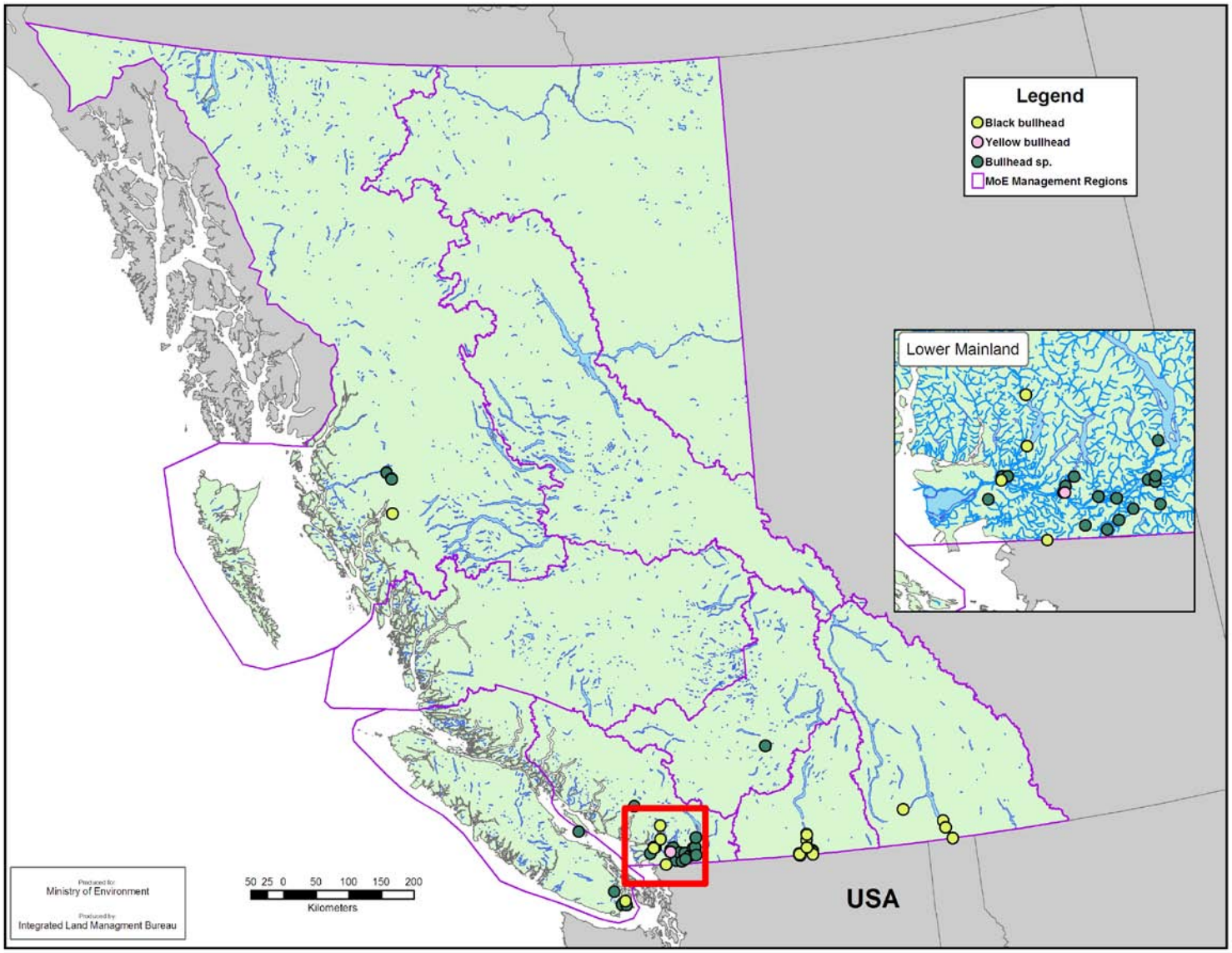


Figure 14. Black catfish occurrences in British Columbia, based on the MoE occurrences database. Note that some specimens were not identified to species, and are tracked simply as “catfish – general.”

or cannot distinguish between the two bullhead species currently found in BC. We assume that at least some of these observations are of black bullhead.

Region	No. of waterbodies with "black bullhead"	No. of waterbodies with "catfish - general"	No. of waterbodies with one or both
1	1	4	5
2	2	18	20
3	-	1	1
4	4	-	-
5	-	-	-
6	1	2	4
7	-	-	-
8	6	-	6
9	-	-	-

Distribution Trends.— Data indicate a steady spread of this species in BC.

	Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
black bullhead	1	3	11	14
catfish - general	1	4	11	25

Stocking Records for BC.— There are no records of authorized stocking of black bullhead in British Columbia.

Mode of Invasion.— Black bullhead have been intentionally stocked for food and sport and have generally established in most waters where introduced (Fuller 2005a). The species has likely dispersed naturally to downstream areas where habitats exist, and upstream where gradients are low and streamflows are gentle. Upstream dispersal is cited as the mode of invasion into southern British Columbia from neighbouring states (Scott and Crossman 1973), and many of the observed locations reflect this (Figure 14). However, the observations on Vancouver Island suggest illegal introductions directly into these waterbodies. The extent to which the species is still being introduced is not known, but is presumably not great due to the species' limited appeal (at least within Canada) as a sportfish. Given that this is a warm-water species , it is not surprising that most observations are limited to southern areas of BC, and it is unlikely that this species has the potential to spread into cooler, northern areas of the province under current temperature conditions.

6.5.7 Threats

Threats from black bullhead to native fish species have not been quantified or described in detail, but are likely similar to those from brown bullhead. The species is listed as a common fish pest (FishBase 2005). Because it is a nocturnal benthic feeder, black bullhead may be considered a potentially serious predator of eggs for any littoral spawning species; since black bullheads are opportunistic omnivores they may also be a direct threat to small fish (Fuller

2005a). Benthivorous fishes in general have been implicated in the increased turbidity of shallow lakes and ponds, primarily through the removal of soil-stabilizing macrophytes (Braig and Johnson 2003). Changes in turbidity, if severe, can affect aesthetics, water quality, and fish and wildlife communities (Zambrano 2001). In the US, black bullheads are considered a pest species where they have been introduced. Fuller et al. (1999 and references therein) summarize the following impacts. In Arizona, black bullhead have produced abundant stunted populations that compete with more desirable fish species for food and space. The species is known to prey on the endangered humpback chubs (*Gila cypha*) in the Little Colorado River, and is a voracious predator of newly hatched gamefish. They may also be partially responsible for the decline of the Chiricahua leopard frog (*Rana chiricaheunsis*) in southeast Arizona.

6.5.8 Potential Management Actions

Black bullhead are very hardy and are considered extremely difficult to eradicate. Scott and Crossman (1973) report that black bullheads are even more tolerant of pollutants and environmental extremes than brown bullheads. Given the general hardiness of bullheads and their resistance to pollutants and piscicides, an effective control or eradication program may require the use of several of the methods discussed in Section 5.

6.6 Black Crappie (*Pomoxis nigromaculatus*)[2]



6.6.1 Taxonomy

Family: Centrarchidae (Sunfishes)
Genus: Pomoxis
Species: nigromaculatus
Authority: Lesueur, 1829

6.6.2 Habitat

Black crappie are found in clear, quiet, warm water of ponds, lakes, bays and shallow areas of lakes, and slow waters in larger rivers (Scott and Crossman 1973). Within these areas they are usually associated with abundant aquatic vegetation and sandy to silty bottoms (Scott and Crossman 1973).

Spawning Habitat.— Males build nests in sand, gravel or mud in 25 to 61 cm of water where there is some vegetation (Scott and Crossman 1973). At times nests are built in the protection of an undercut bank (Scott and Crossman 1973).

6.6.3 Diet

Juvenile Diets.— Young black crappies are largely planktivorous, feeding in the open water column on crustacea and insect larvae (Scott and Crossman 1973).

Adult Diets.— As fish increase in size they become increasingly piscivorous (Scott and Crossman 1973), but may continue to eat a large number of invertebrates (Bonar et al. 2005). Black crappie are taken in an ice fishery, implying that they are active year round and will feed in winter (Scott and Crossman 1973).

6.6.4 Life History

Reproduction.— Males provide all parental care. In the late spring, they acquire territories in the littoral region where they build nests and mate (sometimes with many females). Spawning commences when water temperatures reach 19 to 20 ˚C, usually in late May to mid-July in

[2] Image used with permission from Robert Rice, www.nativefish.org

central Canada (Scott and Crossman 1973). Males guard their nests, fan eggs in the nest and protect juveniles for a short time after hatching (Scott and Crossman 1973).

Age at First Reproduction.— Sexual maturity is attained when these fish reach 2 to 4 years (Scott and Crossman 1973).

Life Span.— Scott and Crossman (1973) list maximum age as 8 to 10 years. FishBase (2005) lists a maximum recorded age of 15 years.

Body Size.— FishBase (2005) indicates a maximum length of 49.0 cm and weight of 2,720 g. Scott and Crossman (1973) suggest that a maximum length of 30 to 36 cm and ~ 1 kg is normal for Ontario.

Fecundity.— Egg number for females 195 to 230 mm ranged from 26,700 to 65,520 with an average of 37,796 (Ulrey et al. 1938 cited in Scott and Crossman 1973).

6.6.5 Human Interest

Black crappie are a commonly targeted sport fish both in their native and non-native ranges, though in BC they do not typically have the high status of many other sport fishes. The species is listed as a game fish in the BC freshwater fishing regulations, with angling restrictions imposed in regions where they exist (British Columbia Government 2005). They can be caught still-fishing with worms or bait. Black crappie has also formed a component of the freshwater commercial fishery in central Canada and elsewhere (Scott and Crossman 1973).

6.6.6 Distribution

Native Distribution.— The natural distribution of black crappie is freshwaters (and occasionally brackish water) of eastern and central North America, but this species has been introduced so widely that the natural distribution is difficult to determine precisely (Fuller 2005). In Canada the species occurs naturally in southern Manitoba, Ontario and Quebec, in the Red and Assiniboine rivers, the Great Lakes and tributaries, and the St. Lawrence and its tributaries downstream to brackish water (Scott and Crossman 1973). The species is naturally absent from the Maritimes, the New England states, and the western half of North America (Scott and Crossman 1973).

Present Distribution.— In addition to its natural distribution in Canada, the species is present in British Columbia and Saskatchewan (NatureServe 2005). It is present in every state except Alaska and Hawaii (Fuller 2005), and is listed as introduced to Mexico and Panama (FishBase 2005). Black crappie are widely distributed in Puget Sound streams and through much of the Columbia system in Washington State (Wydoski and Whitney 2003).

BC Distribution.— The following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. Note that since some waterbodies span more than one MoE region, the total number of unique waterbodies in the province may be different than the sum of counts within each region. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

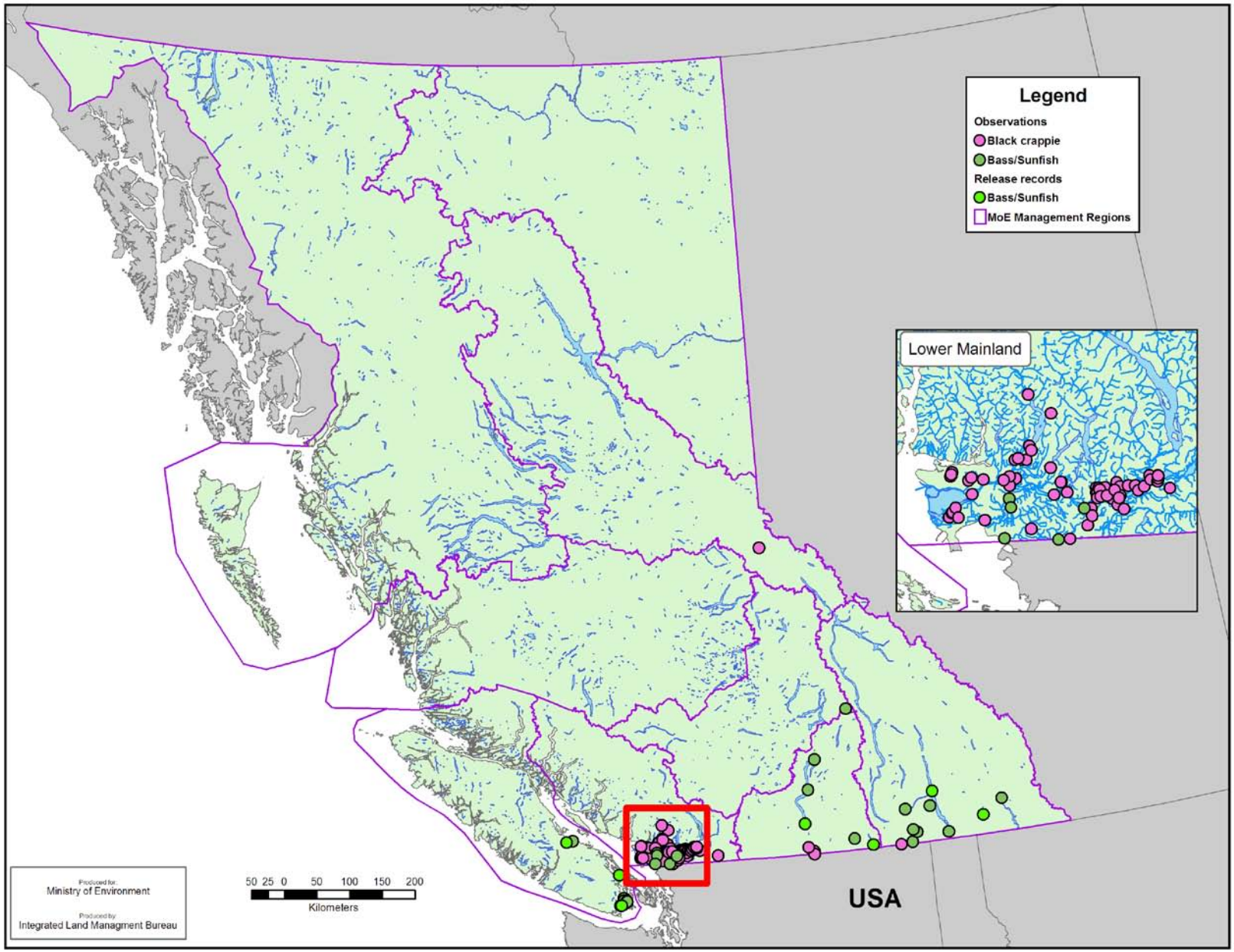


Figure 15. Black crappie occurrences in British Columbia, based on the MoE occurrences database. Note that some centrarchid specimens were not identified to species, and are tracked simply as “bass / sunfish.” It is unlikely, but possible that some of these records are black crappie.

In BC, the distribution of black crappie is primarily restricted to the lower Fraser and its tributaries in the Lower Mainland. A small number of observations have been made elsewhere in the province.

Region	No. of waterbodies
1	-
2	38
3	-
4	1
5	-
6	-
7	1
8	2
9	-

Distribution Trends.— The data indicate a considerable rise in the number of waterbodies with black crappie. Most of this spread has occurred in the Lower Mainland area, and there has been a steady increase in the number waterbodies through time.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
3	16	33	41

Stocking Records for BC.— There is no record in the releases database of authorized stocking of black crappie in BC.

Mode of Invasion.— In the US, black crappie has been widely and successfully introduced through authorized and unauthorized stocking for the purposes of increasing angling opportunities. In Canada, the species has been spread via unauthorized introductions, presumably by anglers seeking to improve angling opportunities. The species was commonly introduced into Washington and Oregon states early in the 20th century, and it is likely that these populations were the source for introductions to BC (Royal British Columbia Museum 2002). Although popular with anglers in many locations where the species is indigenous, black crappie do not usually have the status of trophy fish in BC, so there may be less pressure to translocate this species here. It seems likely that black crappie have spread in the lower Fraser River and its tributaries at least in part through natural dispersal. Given its requirement for warmer temperatures, the spread of this species will likely be limited mainly to warm waters of southern British Columbia.

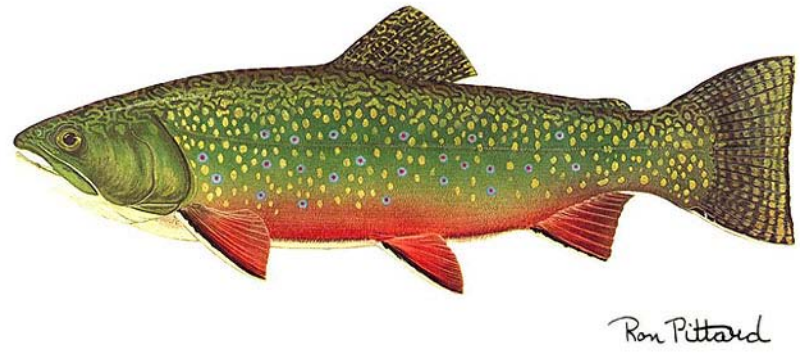
6.6.7 Threats

We found no published information regarding impacts from black crappie to local aquatic biodiversity. Given the widespread distribution of this species beyond its native range, and the fact that negative effects are seldom reported, impacts to native biodiversity are likely not severe.

6.6.8 Potential Management Actions

Section 5 discusses the various methods available for controlling fish species once they are established. However, as with other highly prolific non-native species, black crappie are difficult to eradicate once established. Efforts should therefore be made to prevent dispersal of this species into new habitats, including dissemination of general educational information on the ecological dangers of translocating wildlife.

6.7 Brook Trout (*Salvelinus fontinalis*)



6.7.1 Taxonomy

Family: Salmonidae (Salmonids)
Genus: Salvelinus
Species: fontinalis
Authority: Mitchill, 1814

6.7.2 Habitat

Brook trout have the most generalized habitat requirements of all *Salvelinus* species: they inhabit small headwater streams, large rivers, ponds, and large lakes in inland and coastal areas (Raleigh 1982). Typical brook trout habitats are those associated with a cold, temperate climate, cool, spring-fed groundwater, and moderate precipitation (MacCrimmon and Campbell 1969 cited in Raleigh 1982). Warm water temperatures appear to be the single most important factor limiting brook trout distribution and production (Creaser 1930 cited in Raleigh 1982; Mullen 1958 cited in Raleigh 1982; McCormick et al. 1972 cited in Raleigh 1982).

Optimal brook trout riverine habitat is characterized by clear, cold spring-fed water; a silt-free rocky substrate in riffle - run areas; an approximate 1:1 pool to riffle ratio with areas of slow, deep water; well-vegetated stream banks; abundant instream cover; and relatively stable water flow, temperature regimes, and stream banks (Raleigh 1982). Brook trout south of Canada tend to occupy headwater stream areas, especially when rainbow and brown trout are present in the same river system (Webster 1975 cited in Raleigh 1982). They tend to inhabit large rivers in the northern portion of their native range (Behnke 1980 cited in Raleigh 1982). Optimal lacustrine habitat is characterized as clear, cold lakes and ponds that are typically oligotrophic (Raleigh 1982).

Cover is recognized as one of the basic and essential components of trout streams, and is a major limiting factor for brook trout in streams (Raleigh 1982). The main use of summer cover is probably for predator avoidance and resting. In some streams, the major factor limiting salmonid densities may be the amount of adequate overwintering habitat rather than summer rearing habitat.

Brook trout spawning occurs most often in gravel beds of shallow headwater streams (Scott and Crossman 1973). Brook trout may also spawn successfully in upwelling littoral areas of lakes (Webster 1975 cited in Raleigh 1982).

6.7.3 Life History

Reproduction.— Brook trout are "fall spawners," spawning in late summer or autumn, depending on latitude (Scott and Crossman 1973). Spawning is typical of salmonids. Females excavate a redd in gravel, deposit eggs, which are fertilized by a male, then covered with gravel by the female. This cycle may be repeated several times, with new excavations occurring immediately upstream of the redd. Eggs incubate in the gravel, with no parental care beyond the territoriality that is often exhibited during the mating cycle. Eggs hatch after about 100 days at 6.1 ˚C or 75 days at 10 ˚C (Scott and Crossman 1973). Larvae remain in the gravel until they have absorbed their yolk sac, at about 38 mm in length (Scott and Crossman 1973).

Age at First Reproduction.— Brook trout usually reach sexual maturity at 3 years (Scott and Crossman 1973), but reproduction may occur at 1 or 2 years in some populations (FishBase 2005), or even occasionally at 0+ (Buss and McCreary 1960 cited in Raleigh 1982; Hunt 1966 cited in Raleigh 1982).

Life Span.— Unlike other chars, brook trout are relatively short-lived, seldom exceeding 5 years (Scott and Crossman 1973). The maximum reported age is 7 years (FishBase 2005). However, Raleigh (1982) discusses two forms of brook trout: a small (200 to 250 mm), short-lived (3 to 4 years) form typically found in small cold streams and lakes, and a large (4 to 6 kg), piscivorous form, usually associated with large lakes, rivers and estuaries. The larger form is said to occur primarily in the northern portion of its native range.

Body Size.— Maximum recorded size is 86.0 cm SL and 9.39 kg (FishBase 2005). Fish over 4.5 kg are rare (Scott and Crossman 1973). High density and stunting is common in brook trout populations.

Fecundity.— Brook trout eggs are large — 3.5 to 5 mm in diameter (Scott and Crossman 1973) — so fecundity is fairly low. Marchetti et al. (2004a) list fecundity at 100 to 1000 eggs per female. Paul et al. (2003) used the following fecundity-length relationship: egg number = 57 × FL [mm] – 819, which translates into a fish of 300 mm having a fecundity of 16,281 eggs.

6.7.4 Diet

Brook trout are predatory, and consume a wide variety of organisms, usually opportunistically.

Juvenile Diets.— Young brook trout eat aquatic invertebrates (e.g., insect larvae) and terrestrial insects that drift into or onto water.

Adult Diets.— Larger brook trout eat a wide variety of organisms, including aquatic and terrestrial invertebrates. Where opportunities exist, they will also consume small mammals and other fish. Brook trout engage in some cannibalism, eating both eggs and young brook trout (Scott and Crossman 1973).

6.7.5 Human Interest

This salmonid is a commonly targeted sport fish and was originally transferred to BC waters to enhance angling opportunities. The earliest recorded authorized release of brook trout in BC was in 1908, and the species continues to be released in authorized stocking programs. However, a draft provincial Brook Trout Stocking Policy was developed in 1999 to address growing concerns regarding impacts to native diversity. This policy includes the following: all stocked fish must be triploid, all receiving systems must be considered 'closed' (i.e., no inlet or outlet stream to permit spread) and no new systems will be stocked. This policy has been followed since that time, and all releases have involved only triploid all-female stock.

6.7.6 Distribution

Native Distribution.— Scott and Crossman (1973) describe brook trout as endemic to northeastern North America. Within Canada, brook trout are naturally widely distributed in Manitoba, Ontario, Quebec, the Maritimes, and Labrador and Newfoundland. Its distribution extends northward to about 60˚ in many systems draining into Hudson Bay and along the Atlantic coast. In the northeastern United States, brook trout naturally occur predominantly around the Great Lakes region, the New England states and inland in the Appalachian Mountains.

Present Distribution.— Brook trout were originally introduced to western North America via human effort, but have subsequently spread over large distances through unaided dispersal (Dunham et al. 2002). Brook trout have been successfully introduced to many temperate regions throughout the world, including Europe, Asia, South America, southern Africa, Australia, New Zealand, and many islands (Scott and Crossman 1973; FishBase 2005). Brook trout are widely distributed in the northern interior of Washington State (Wydoski and Whitney 2003) and other regions near the BC border.

BC Distribution.— Figure 16 and the following table summarize unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. Note that since some waterbodies span more than one MoE region, the total number of unique waterbodies in the province may be different than the sum of counts within each region. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1. Note that splake (brook trout × lake trout hybrids) were reportedly stocked into a limited number of lakes for a short period.

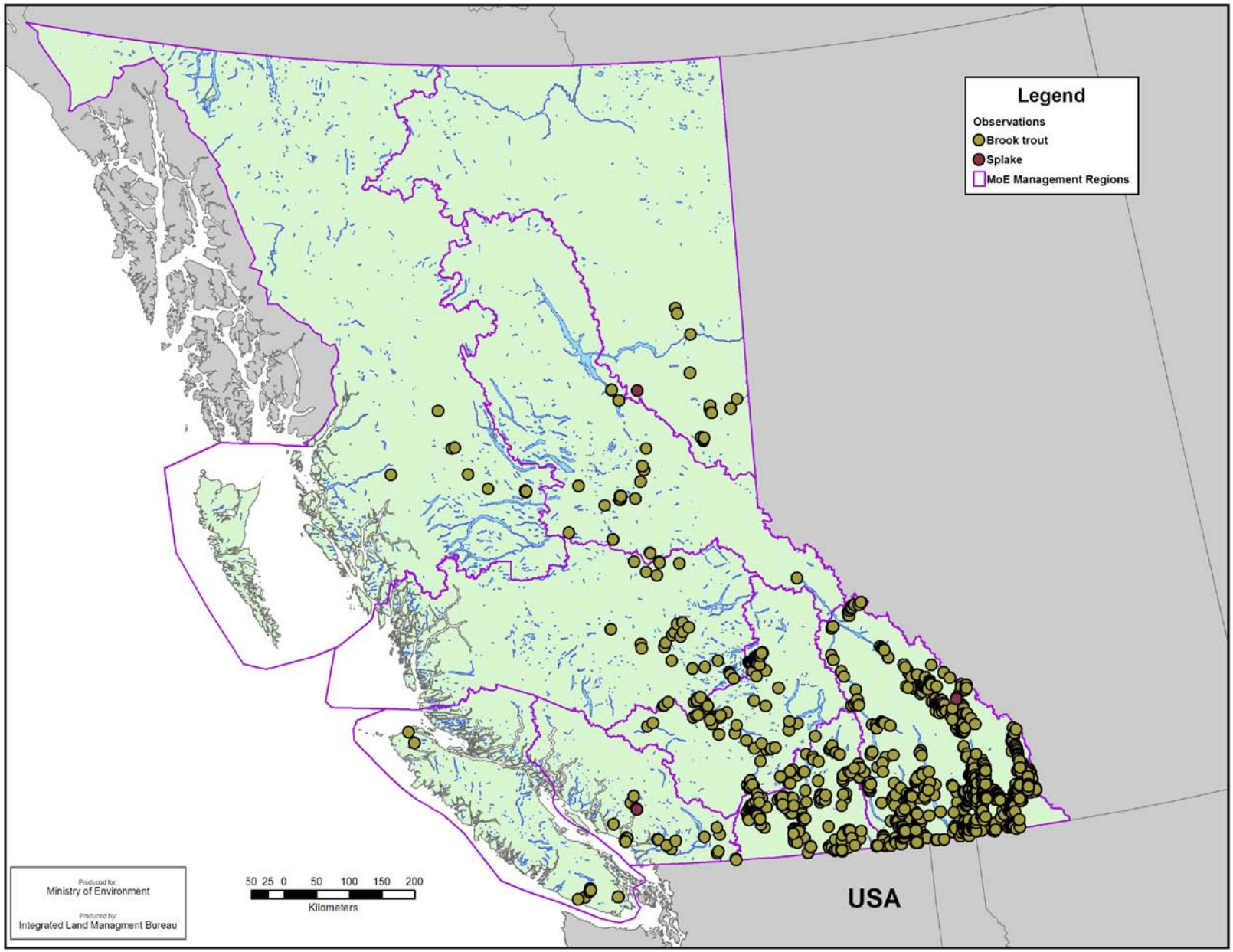


Figure 16. Brook trout occurrences in British Columbia, based on the MoE occurrences database.

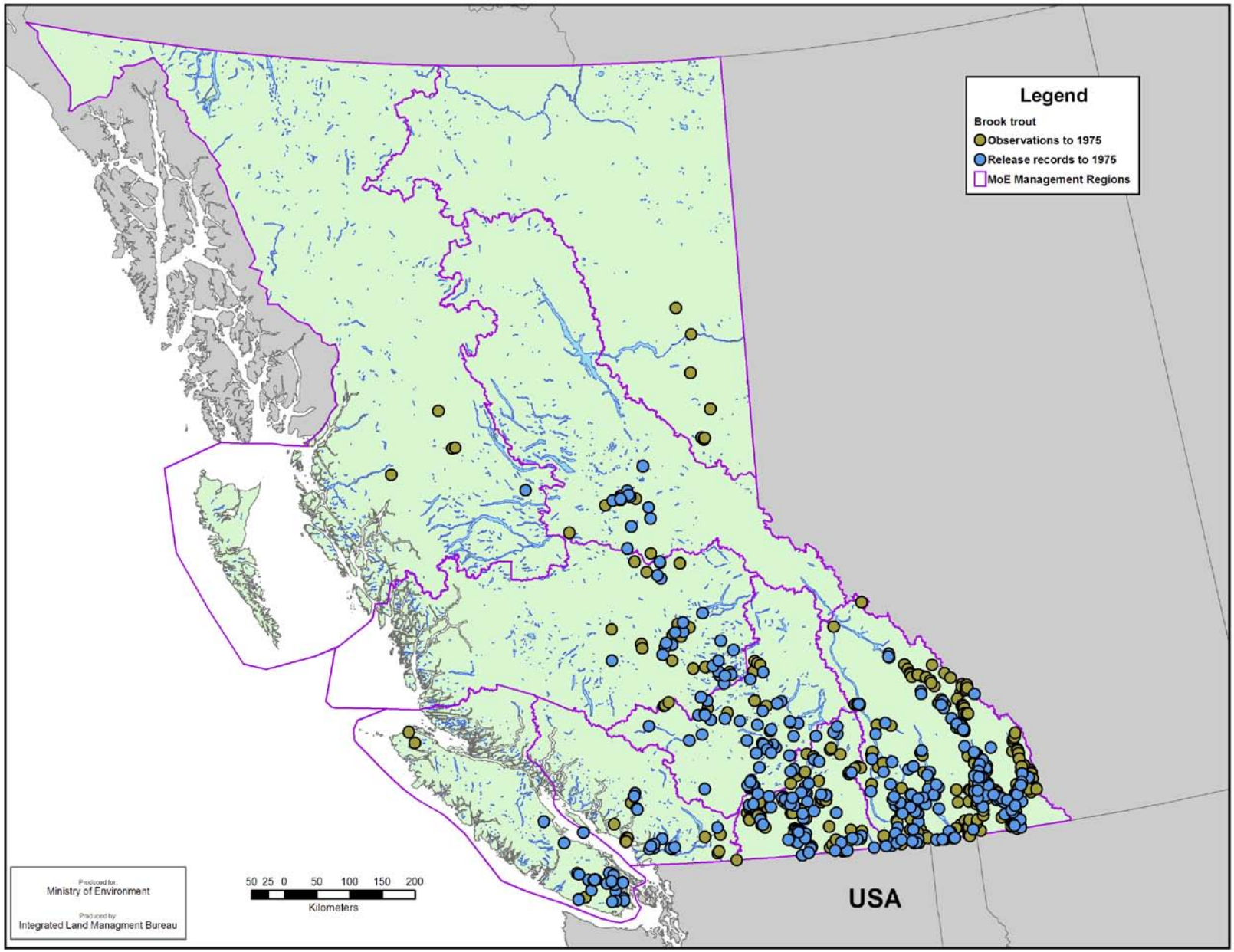


Figure 17. Pre-1976 stocking locations of brook trout in British Columbia, based on the MoE releases database.

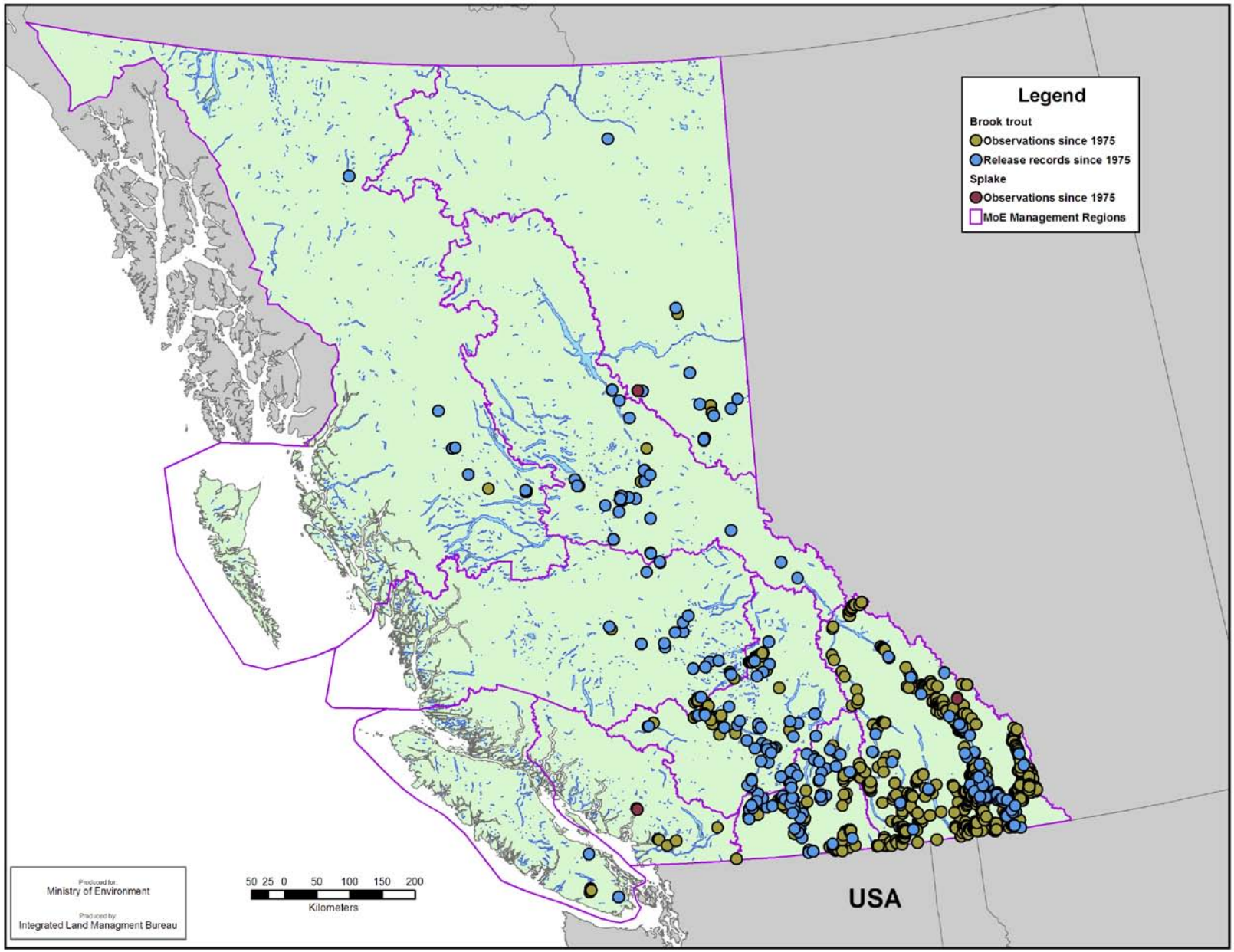


Figure 18. Post-1975 stocking locations of brook trout in British Columbia, based on the MoE releases database.

Region	No. of waterbodies
1	5
2	16
3	47
4	402
5	26
6	8
7	22
8	132
9	12

Brook trout are not native to British Columbia, but they have been recorded in all MoE regions of the province, particularly in the Kootenay and Okanagan regions. Occurrence records for brook trout date back more than 60 years, but it is not known in how many of these waterbodies brook trout have gone extinct, nor do we know in how many unsurveyed waterbodies brook trout currently exist. These data nevertheless imply a very broad distribution within BC, and concur with Taylor's (2004) finding that brook trout are the most widely distributed non-native fish species in BC. It should be noted that these observation data do not distinguish between systems that have and have not been stocked under authorized means with brook trout; the majority of systems with observations have either been stocked directly, or stocking has occurred in the same watershed. Notable exceptions include the Nahwitti system on north Vancouver Island, Lakelse Lake in Region 6, Chapman Lake (Sechelt area), a Bowen Island observation and Ross Lake in the Skagit watershed. The Nahwitti observations are possibly mis-identified Dolly Varden (S. Rimmer, pers. comm.). This may also be the case for Chapman Lake and Bowen Island. Ross Lake is likely the result of fish moving north from Washington State, where they have been introduced. Established populations have been confirmed for only two systems on Vancouver Island including Dimple Lake and Spectacle Lake (S. Rimmer, pers. comm.); both are associated with earlier stocking efforts.

Distribution Trends.— Data indicate an increasing trend in the number of waterbodies in BC with brook trout. The trend toward increasing numbers of waterbodies with brook trout is likely accurate to some degree, but care should be exercised when using these data to estimate rates of change in the species' distribution (see discussion of database in Section 6.1). For example, biological surveys have not been conducted systematically in time and space, so in part these data simply reflect a cumulative increase in observations over time.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
12	89	527	668

Brook trout were likely established in many waterbodies prior to 1950, since stocking records date back to the early 1900s in many MoE regions. The dearth of early records likely means that surveys had either not been conducted or the records were not incorporated into the MoE

database, rather than an absence of established brook trout populations. As noted earlier, it is not known in how many waterbodies brook trout have gone extinct, nor do we know in how many unsurveyed waterbodies brook trout currently exist. Note also, that some records in the database do not have years associated with the record, so the total number of waterbodies exceeds the value in 2004, when the last survey record was posted for this species. Despite shortcomings, the data emphasize the widespread distribution of brook trout within BC, and an apparent increase in the range of this species.

Stocking Records for BC.— Brook trout were originally transferred to BC waters to enhance angling opportunities, and have been part of stocking programs in the province since the early 1900s. The species continues to be released in authorized stocking programs, but all stock is now triploid all-female. Records from the releases database indicate that the species has been part of an authorized stocking program in all MoE regions of BC. Stocking of brook trout has been discontinued in MoE regions 1 and 2 due to concerns of impacts to native species and failure of brook trout to thrive (especially Region 1). Overall, the number of stocked systems has declined, as stocking practices became more conservative (i.e., only closed lakes are now stocked, and no new systems are stocked) and fisheries managers became more aware of the potential impacts of this species on native species (Figure 17 and Figure 18).

Region	No. of waterbodies	No. of releases	Years
1	30	128	1908-1990
2	13	15	1908-1980
3	61	977	1911-2005
4	143	1364	1911-2005
5	27	279	1929-2005
6	6	118	1963-2005
7	30	375	1960-2005
8	84	1024	1916-2005
9	11	120	1978-2005

Mode of Invasion.— Brook trout were originally introduced to BC through an authorized stocking program in efforts to enhance angling opportunities. Authorized stocking continues today in many regions of the province although stocking practices are now more conservative and use only triploid all-female stock. The extent of unauthorized stocking, both past and present, is not known but is likely not a substantial component of the establishment and spread of brook trout in BC. There is also one example of a small lake in Region 7 receiving reproductive brook trout mistakenly (it was mis-identified by hatchery staff). It is not known how many other similar occurrences there may be.

Natural dispersal after stocking has almost certainly increased the range of brook trout in BC, though likely to different extents in different areas. When fish are stocked in headwaters there are fewer barriers to downstream dispersal in comparison to upstream dispersal, and even very steep slopes and vertical barriers can be overcome (Adams et al. 2001). This means that a single successful stocking event in a headwater system can lead to a considerable increase in distribution downstream of the release site. Paul and Post (2001) analysed the spatial distribution of brook trout relative to stocking location in Kananaskis Country, Alberta, and

found a general tendency to colonization downstream of stocking locations. A similar analysis has not been conducted in BC.

Other studies (Gowan and Fausch 1996; Adams et al. 2000; Peterson and Fausch 2003b) have demonstrated an upstream bias in brook trout movements, though a similar bias in colonization need not occur from this movement bias if colonization success differs among upstream and downstream locations. Brook trout are effective upstream dispersers and have been shown to overcome barriers of 22% slope and 1.2m vertical falls (Adams et al. 2000), and to disperse distances of greater than 2000m in a summer (Gowan and Fausch 1996; Peterson and Fausch 2003b). In sum, whether there is an upstream or downstream bias in colonization is somewhat moot: evidence indicates that brook trout are effective colonizers throughout large portions of many watersheds.

6.7.7 Threats

Brook trout have taken over large geographic areas of native salmonid habitat throughout western North America, particularly that of cutthroat trout and bull trout. Brook trout also hybridize with bull trout, resulting in genetic effects in addition to habitat invasion. The patterns and effects of invasion have been best studied with cutthroat trout and bull trout, but effects are assumed to extend to other species also. The precise mechanisms of invasion are not necessarily well-understood, and may vary among streams. There is no published evidence of brook trout impacts within BC, but there seems little reason to doubt that there are similar impacts here. Elsewhere, this fish replaced the native greenback cutthroat trout (*O. clarki stomias*) in a Colorado creek over the short period of only five years (Fuller et al. 1999 and references cited therein). It has also replaced the native Lahontan cutthroat trout (*O. clarki henshawi*) and the golden trout (*O. aguabonita*) in all areas where the species come into contact (Fuller et al. 1999 and references cited therein). In Montana, brook trout inhabit at least 40% of native bull trout streams (Fuller et al. 1999).

A number of studies have shown brook trout range expansion at the expense of native salmonids in the western US, particularly cutthroat trout (reviewed in Dunham et al. 2002) and bull trout (Leary et al. 1993). Declines in all seven subspecies of inland cutthroat trout were apparent starting in the late 19th century and are continuing today (Dunham et al. 2002). Causes of the decline have been attributed to habitat loss and degradation, harvest, and influences of artificial propagation. The expansion of brook trout distribution has been heavily implicated in native species decline and has motivated a considerable research effort toward understanding the effect of brook trout invasions.

Patterns of segregation are clear: brook trout distribution has increased while the distributions of native species like cutthroat trout have declined. What remains unclear is the extent to which brook trout have *displaced* or *replaced* native species (Dunham et al. 2002). Displacement results from biotic interactions between species, where brook trout actively outcompete, introduce diseases to, or directly consume the native species. Replacement on the other hand occurs when a native species declines for reasons unrelated to the presence of brook trout (e.g., declines in habitat quality, or overharvest). Brook trout then advance in their distribution because they are better able to withstand the disturbances than the native species. The question of displacement vs. replacement is of significant management concern, because the answer can influence

decisions about how best to manage native species and the invasion of brook trout. For example, if replacement is occurring following overharvest of the native species then the best management solution may be to alter fishing access and regulations. On the other hand, if displacement is occurring then managers may wish to consider the feasibility of eradication of brook trout.

Many of the studies of brook trout invasion to date have been descriptive rather than experimental, primarily describing the pattern of species segregation or the habitat use patterns (e.g., Shepard 2004). Virtually all of these are in agreement, brook trout now occupy an enormous number of lowland streams and some native salmonids are restricted to high elevation areas where brook trout are absent (Paul and Post 2001; Dunham et al. 2002). Many other studies have focused on biotic interactions at the individual level—often assuming displacement rather than replacement (e.g., Hilderbrand and Kershner 2004). These studies are often equivocal, showing competitive advantages for brook trout in some situations but not all (e.g., Gunckel et al. 2002; see also reviews in Dunham et al. 2002).

Dunham et al. (2002) review evidence for a number of hypotheses regarding the mechanisms of invasion of brook trout into cutthroat trout areas including, temperature tolerance, stream slope, competition, predation, disease, angling pressure, disturbance, and stocking location. There is a varying degree of support for each of these mechanisms and none seem to be universal. They conclude, perhaps not surprisingly, that the evidence for displacement is often lacking and that brook trout do not appear to pose an unconditional risk to cutthroat. In some cases brook trout invade rapidly, whereas in other cases cutthroat seem able to resist invasion over long periods of time (Adams et al. 2002). Their results imply that different mechanisms may dominate in different brook trout invasions.

More recent studies of brook trout invasion in Colorado streams demonstrated a very clear effect of brook trout presence on cutthroat trout growth and reproduction (Peterson and Fausch 2003a, b). A study by Levin et al. (2002) in the Columbia River basin showed that chinook salmon juveniles in streams without brook trout had nearly double the survival of juveniles in streams with brook trout. Although the mechanisms operating at the individual-level are unknown, these recent studies are some of the best to date in that they demonstrate population-level effects of brook trout presence and clearly demonstrate a strong biotic force by brook trout on native salmonid distribution. In part, this may be due to early maturation of brook trout in comparison to most western salmonid species, and their high intrinsic rate of production.

At one level the discussion of invasion mechanisms and displacement vs. replacement is of interest scientifically, and is especially important where discussions are taking place about a specific location for restoration or eradication. But at another level the discussion misses an important point: the invasion of brook trout has been successful over a very broad geographic scale even in areas where disturbances of all kinds are low, such as in parks. This implies that brook trout can invade over a broad range of conditions, conditions that are within the normal range of variation for most areas. Considerable care should therefore be taken to protect those areas that currently do not have brook trout.

In addition to risks of habitat invasions, brook trout also pose genetic risks for closely related species, such as bull trout. Brook trout and bull trout are congeners with no overlap in their

natural distribution, but with extensive secondary contact following the introduction of brook trout in western North America. Genetic evidence indicates that brook trout hybridize with bull trout over a wide geographic range in the western US (Leary et al. 1993). Using diagnostic allozyme loci and mitochondrial DNA, Leary et al. (1993) showed that many individuals in the zone of contact are of hybrid origin. Furthermore, the hybrids were shown to be F1 hybrids of bull trout females and male brook trout. The absence of F2 and backcross hybrids indicated that F1s were sterile or selected against during courtship. Because F1s were more commonly from bull trout females, hybridization was thought to represent a greater waste in reproductive effort for bull trout than brook trout, and hence aid the invasion of brook trout into bull trout areas. Subsequent work (Spruell et al. 2001; Kanda et al. 2002) has shown that introgression does occur in the wild (i.e., that F1s can and do reproduce). Although it is not documented in the literature, it seems possible that brook trout will hybridize with Dolly Varden, another congener, where they co-occur.

Ironically, within the native range of brook trout they are often in decline and are sensitive to introductions of rainbow and brown trout (Raleigh 1982; Marschall and Crowder 1996).

6.7.8 Potential Management Actions

The widespread distribution of brook trout in BC, their historic and ongoing use in fish stocking programs, and the perceived public benefits of the species mean that management of brook trout may be complex and require different approaches in different locations. Indeed, in some locations there may be conflicting objectives between conservation of native diversity and management of recreational angling, and it may require some effort to determine the precedence of these objectives and whether they can reasonably coexist. A serious examination of stocking policies and practices would include an assessment of direct and indirect risks to biodiversity in BC. Direct risks from stocking would arise from release of sexually viable stocks into new waterbodies, or those that do not have a clear and effective barrier to dispersal. Indirect risks may arise if it becomes difficult for the public to differentiate between different management actions in different circumstances. For example, it may be difficult to educate the angling community that this is a "bad" species and should not be translocated, if managers are actively releasing brook trout in stocking programs elsewhere (even if the stocked fish are triploid). There is likely some risk in sending a mixed message if managers allow stocking and enhancement for some brook trout populations but implement prohibitions or eradications in other areas. In 1999, the Ministry of Environment produced a draft Brook Trout Stocking Policy to address some of the above issues, in particular reproduction and spread. As a result, only triploid all-female juveniles are now stocked, no new systems (i.e., those that have not received brook trout in the past) are stocked and all lakes must be "closed" (i.e., no apparent means of escape). This draft has not been finalized but is nevertheless adhered to. Unfortunately, it does not address existing established populations of brook trout.

Options for direct control of brook trout abundance and distribution fall into the two classes of 1) preventing further spread of the species and 2) eradication or population reduction in areas where brook trout currently exist. Preventing further spread of the species is in theory straightforward through the construction of dispersal barriers. Adams et al. (2000) note several projects that have created bull trout refugia through construction of barriers to prevent upstream migration of brook trout. But this approach isolates the target native species as well

as the invader, and runs some risk that migration of the native species is required to maintain the integrity of that population. Other concerns include the fact that such technology is likely only feasible on relatively small streams, and geomorphic processes of the stream may require a very robust (and therefore expensive) structure. And, finally, the jumping capabilities of brook trout need to be well-understood to ensure that any structure built is indeed a barrier to upstream dispersal (Adams et al. 2000).

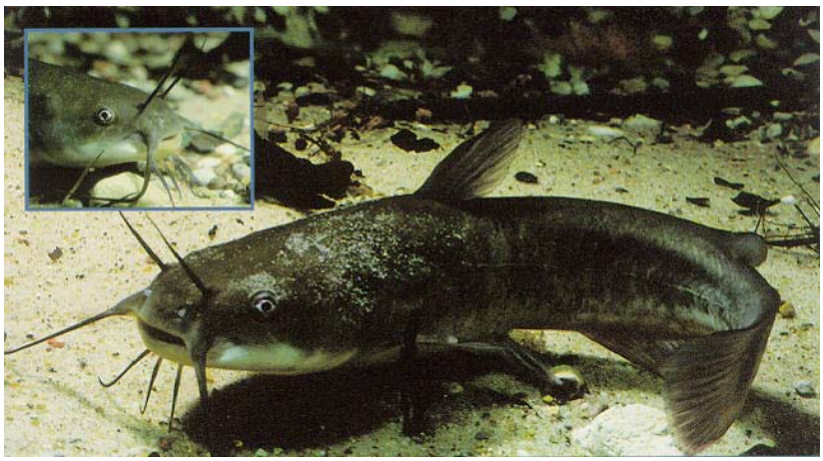
Options for eradication or abundance control are also of fairly limited scope. Typically it is very difficult and expensive to remove any species once it is established, and except for a very few locations this is likely to be also true for brook trout. Gill-netting, trapping and electrofishing are generally not effective for complete removal, and brook trout will reinvade unless treatments are sustained. But the methods can be used to lower population abundance to a level that will confer an advantage to a non-targeted native species (Peterson et al. 2004). Another approach is the use of toxins, such as rotenone and antimycin. These usually require multi-year, intensive treatments (Dunham et al. 2002). Once brook trout are eradicated, the target species is re-introduced or allowed to recolonize from untreated sources, and a dispersal barrier installed to prevent re-invasion by brook trout. The downside of this method is that the toxins are not species-specific, so there is the potential of unintended side-effects, and use of toxins may be met with public resistance particularly if used in settled regions. A directed fishery on Quirk Creek in Alberta is one example where angling is being used as a method of eradication of brook trout in a native bull trout and cutthroat trout stream. This project has been ongoing for several years and has required significant funds to coordinate angling efforts and monitor progress. It is not yet clear how successful this program will be, but modelling suggests that due to the resiliency of brook trout and relatively high catchability of bull trout, this management technique may not succeed (Post and Johnston 2002).

Use of fishing regulations to favour selective harvest of brook trout seems a tempting approach, but is not likely to be effective. Brook trout are a highly-regarded and targeted sportfish, but they can be more difficult to catch than native species (e.g., catchability of cutthroat trout is ~3 x higher than for brook trout; Paul et al. 2003), meaning that native species may suffer substantial hooking mortalities even after release. Paul (2000 cited in Dunham et al. 2002) modeled the effects of fishing on cutthroat trout and brook trout in a small stream and found that even catch-and-release fishing with a 10% hooking mortality rate could lead to the extirpation of cutthroat trout in 5 years, whereas brook trout could sustain harvest with no bag limit indefinitely. This is in part due to early maturation of brook trout, which results in small mature fish that are not highly vulnerable to angling. One final aspect of this policy to consider is that many anglers have difficulty properly identifying fish species (Schmetterling and Long 1999), which may result in higher than expected harvest for non-target species.

Managers should also properly consider that no action may be appropriate in cases where there is no threat to non-native species and low likelihood of further spread. In this, and in all cases, it is required to have a good scientific basis for management decisions. For example, an understanding of the mechanisms of invasion are required to make a defensible decision because it is necessary to distinguish between species replacement and species displacement for species invasions. Direct or indirect control of brook trout may be warranted if there is evidence of species displacement. On the other hand, if brook trout are merely replacing native

species that are declining for some other reason (e.g., harvest, changes in habitat quality, etc.) then effort may be better expended addressing that issue.

6.8 Brown Bullhead (*Ameiurus nebulosus*)[3]



6.8.1 Taxonomy

Family: Ictaluridae (North American freshwater catfishes)
Genus: Ameiurus
Species: nebulosus
Authority: Lesueur, 1819

6.8.2 Habitat

Brown bullhead is a warm-water species that occupies lakes and sluggish streams with muddy or weedy beds (Barnes and Hicks 2003). They are occasionally recorded in estuarine waters (Scott and Crossman 1973). They mostly utilize relatively shallow (< 10m) littoral areas for feeding, and areas of vegetation for cover (Dedual 2002). They are tolerant of a wide range of environmental conditions that may be limiting for other fish species, and can survive temperatures as high as 36˚ C and oxygen levels as low as 0.2 ppm (Scott and Crossman 1973). They do better than many species in polluted waters (Scott and Crossman 1973; Davis et al. 2003).

Moyle and Nichols (1973) and Brown and Moyle (1997) found that non-native brown bullheads have been highly successful in California in reservoirs and in highly regulated streams, where they are often a dominant species. During a long-term study on the Eel River, abundance of brown bullheads was inversely correlated with "naturalness" of the flow regime: they did well in the relatively calm, warm waters common during periods of drought, but were distinctly less abundant during periods when more natural, cooler peak flows occurred (Brown and Moyle 1997).

There is no mention in the literature of differential habitat use by juveniles and adults.

[3] Image used with permission. Copyright Steffen Zienert.

Spawning Habitat.— Brown bullheads build nests and spawn in shallow littoral areas of lakes and creeks mouths (Scott and Crossman 1973). They prefer mud or sand substrate, usually near some cover (such as vegetation, root clumps, or even refuse such as boards or tires, Scott and Crossman 1973).

6.8.3 Diet

Brown bullheads are nocturnal, omnivorous, bottom feeders.

Juvenile diets.— Juveniles feed predominantly on benthic insect larvae such as chironomids and mayflies, as well as cladocera, ostracods, and amphipods (Scott and Crossman 1973). As individuals grow they consume larger items including small fishes and eggs (Scott and Crossman 1973).

Adult Diets.— Adults consume foods similar to juveniles, but may additionally consume molluscs, fish, crustaceans, insects, detritus, plant material, sewage, and other waste (Scott and Crossman 1973; Barnes and Hicks 2003).

6.8.4 Life History

Reproduction.— Brown bullheads spawn in late spring and summer when water temperatures increase to around 21˚ C (Scott and Crossman 1973). Repeated spawning through September may occur in the southern portions of the distribution (Scott and Crossman 1973). A shallow nest is excavated by one or both parents in mud or sandy substrate near the cover of logs, rocks, or vegetation, usually in water less than 1m (Scott and Crossman 1973). Eggs are laid in an adhesive cluster in the nest site, apparently in daytime (Scott and Crossman 1973). Both parents guard the eggs and aerate them by fanning, physically stirring them up, and taking them into the mouth and spitting them back out (Scott and Crossman 1973). After hatching, the young are shepherded and protected by one or both parents until they are about 50mm, after which the juveniles disperse (Scott and Crossman 1973).

Age at First Reproduction.— Females are usually sexually mature by age three (Scott and Crossman 1973; Wydoski and Whitney 2003).

Life Span.— Maximum age is listed in Scott and Crossman (1973) as six to eight years, and is listed in Marchetti et al. (2004a) as nine to sixteen years.

Body Size.— Adults are usually less than 500g and 350mm, though individuals as large as 530mm and 3.6kg have been reported (Scott and Crossman 1973). Wooding (1994) suggests that average weights in Canada are less than 450g, though individuals up to 1.8kg are occasionally caught.

Fecundity.— Scott and Crossman (1973) list fecundity as 2,000 to 13,000 for females of 200mm to 230mm length, though Marchetti et al. (2004a) list maximum fecundity as <10,000 eggs per female.

6.8.5 Human Interest

Brown bullheads are of limited interest to anglers in British Columbia, although they are apparently easy to catch with still-fishing methods and have tasty flesh (Scott and Crossman 1973; Wooding 1994). The species is relatively easy to rear as a pond culture fish, and has been fished commercially in some locations in Canada such as Ontario (Scott and Crossman 1973).

6.8.6 Distribution

Native Distribution.— The brown bullhead is native to the freshwaters of eastern and central North America. It also occurs, rarely, in brackish waters. In Canada, it is found from Nova Scotia, through New Brunswick, the southern tip of Quebec, across southern Ontario at the level of Lake Nipissing, to the north shore of Lake Superior, through southern Manitoba and as far west as the Whitesand-Assiniboine system of southeast Saskatchewan (Scott and Crossman 1973). In the United States, the brown bullhead occurs naturally south to Florida, and inland to Alabama, Mississippi, Louisiana, Oklahoma and Missouri (Scott and Crossman 1973).

Present Distribution.— The brown bullhead was among the first fish introduced to the west coast of the US, as early as 1874 (Fuller 2005b). It has been extensively and successfully introduced in the western US, many European countries, Russia and China (Scott and Crossman 1973; FishBase 2005). Brown bullhead is widely distributed in Puget Sound streams, the lower Columbia River mainstem, and the upper Columbia and Snake watersheds in Washington and Idaho (Wydoski and Whitney 2003).

BC Distribution.— The following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Region	No. of waterbodies with "brown bullhead"	No. of waterbodies with "catfish - general"	No. of waterbodies with one or both
1	20	4	21
2	50	18	62
3	0	1	1
4	4	-	4
5	0	-	0
6	0	2	2
7	1	-	1
8	3	-	3
9	0	-	0

The great majority of observations of brown bullhead come from southern Vancouver Island and the Lower Mainland (Figure 19), in part reflecting its warm-water preferences but also proximity to urban areas and the international border. The database tracks occurrences of black bullhead and brown bullhead separately, but there is also a third category of "catfish – general," for observations that do not or cannot distinguish between the two bullhead species currently found in BC. We assume that at least some of these observations are of brown bullhead.

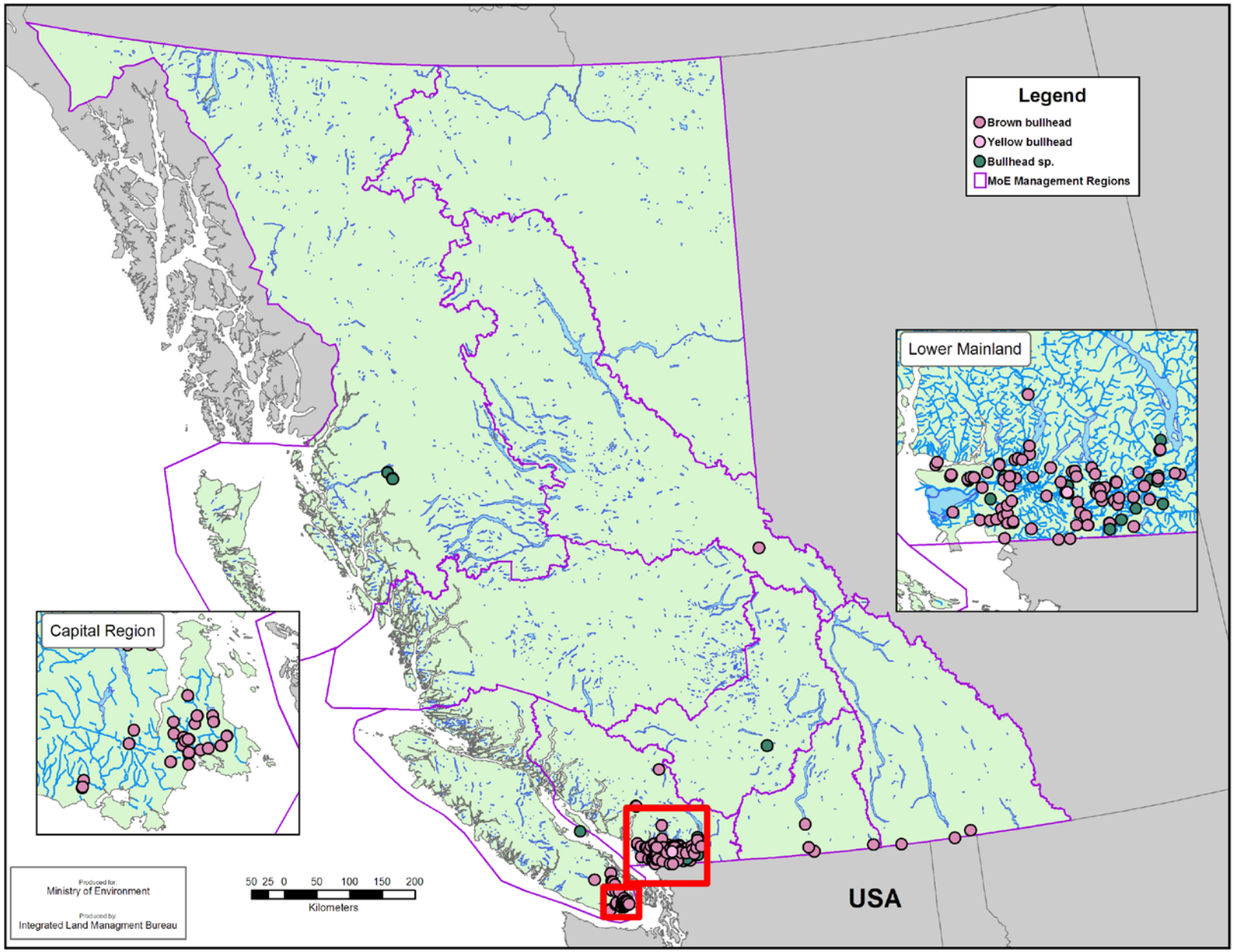


Figure 19. Brown catfish occurrences in British Columbia, based on the MoE occurrences database. Note that some specimens were not identified to species, and are tracked simply as "catfish – general."

Distribution Trends.— Data indicate a steady spread of this species in BC.

	Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
brown bullhead	3	28	57	76
catfish - general	1	4	11	25

Stocking Records for BC.— There are no records of authorized stocking of brown bullhead in British Columbia.

Mode of Invasion.— Brown bullhead have been intentionally stocked for food and sport and have generally established in most waters where introduced (Fuller 2005b). The species has likely dispersed naturally to downstream areas where habitats exist, but likely has not extended its range substantially upstream except where gradients are low and streamflows are gentle. The extent to which the species is still being introduced is not known, but is presumably not great due to the species' limited appeal (at least within Canada) as a sportfish. Proximity to the US border of many observations suggests that some occurrences are due to spread northward from original points of introduction in the US.

6.8.7 Threats

The species is listed as a common fish pest (FishBase 2005b), but threats from brown bullhead to native fish species have not been quantified or described in detail. Its unauthorized introduction to Hadley Lake on Lasqueti Island has been blamed for the extinction of an endemic stickleback species pair, the mechanism being nest predation (Hatfield 2001). When brown bullhead were introduced to other lakes, threespine stickleback were extirpated (McPhail and Lindsey 1986 cited in Wydoski and Whitney 2003). The species has also been flagged as a potential threat to Nooksack dace and Salish sucker, endangered fish found in small, low-gradient streams in the Lower Mainland (Pearson 2004). Because it is a nocturnal benthic feeder, brown bullhead may be considered a potentially serious predator for any littoral spawning species; since brown bullheads are opportunistic omnivores they may also be a direct threat to small fish. Scott and Crossman (1973) suggest that brown bullheads are effective competitors with other benthic-feeding fishes. Benthivorous fishes in general have been implicated in the increased turbidity of shallow lakes and ponds, primarily through the removal of soil-stabilizing macrophytes (Braig and Johnson 2003). Changes in turbidity, if severe, can affect aesthetics, water quality, and fish and wildlife communities (Zambrano et al. 2001).

6.8.8 Potential Management Actions

Brown bullhead are very hardy and considered extremely difficult to eradicate. Scott and Crossman (1973) report that brown bullheads will burrow into mud to avoid adverse conditions, and note their considerable resistance to domestic and industrial pollution. Brown bullheads have tolerances to piscicides that can be more that two hundred times that of salmonids (Govindarajulu 2005) and can apparently survive for weeks lying dormant in the

mud of a dried-up pond. Despite these challenges, brown bullhead has apparently been successfully extirpated from waters in New Mexico (Sublette et al. 1990 cited in Fuller 2005).

Brown bullhead are relatively easy to catch in large numbers using minnow traps and fyke nets, so this may be used as a method to partially control abundance. Given the hardiness of brown bullheads, an effective control or eradication program may require the use of several of the methods discussed in Section 5.

6.9 Brown Trout (*Salmo trutta*)



6.9.1 Taxonomy

Family: Salmonidae (Salmonids)
Genus: Salmo
Species: trutta
Authority: Linnaeus, 1758

6.9.2 Habitat

There are both resident and migratory forms of brown trout, so habitats used vary considerably as individuals make use of streams, lakes and the ocean. Anadromous "sea-run" populations are common in Europe where there is access to the ocean. This form uses freshwater streams for spawning and sometimes for overwintering (Laikre 2000). It is possible that no true anadromous brown trout populations exist in North America, though there are examples of estuarine feeding populations (Behnke pers. comm. cited in Raleigh et al. 1986). Sea-run populations in Europe tend to use inshore habitats (e.g., Jonsson 1985).

Optimal brown trout riverine habitat is characterized by clear, cool to cold water; a relatively silt-free rocky substrate in riffle-run areas; a 50% to 70% pool to 30% to 50% riffle-run habitat combination with areas of slow, deep water; well vegetated, stable stream banks; abundant instream cover; and relatively stable annual water flow and temperature regimes. Brown trout tend to occupy the lower reaches of low to moderate gradient areas (~1%) in suitable, high gradient river systems (Raleigh et al. 1986). Optimal lacustrine habitat is characterized by clear, cool to cold, deep lakes that are typically oligotrophic, but may vary in size and chemical quality, particularly in reservoir habitats.

The upper, near lethal water temperature for brown trout (Needham 1969 cited in Raleigh et al. 1986) is 27.2 ˚C, at which naturally reproducing, viable stream populations would not be maintained. Optimal temperature requirements for good growth and survival of brown trout are 12 to 19 ˚C (Frost and Brown 1967 cited in Raleigh et al. 1986; Mills 1971 cited in Raleigh et al. 1986; Brown 1973 cited in Raleigh et al. 1986; Tebo 1975 cited in Raleigh et al. 1986), with a temperature tolerance range of 0 to 27 ˚C (Maciolek and Needham 1952 cited in Raleigh et al. 1986; Mills 1971 cited in Raleigh et al. 1986).

Spawning Habitat.— Brown trout normally are stream spawners and require tributary streams with gravel substrate in riffle-run areas for optimal reproduction to occur (Raleigh et al. 1986). Spawning habitats are similar for all life history types. Spawning and incubation requires clean gravel substrate with a diameter of about 1.0 to 7 cm, but gravel from 0.3 to 10 cm is useable (Raleigh et al. 1986). The maximum usable gravel size is dependent on the size of the spawning female (Raleigh et al. 1986).

6.9.3 Diet

Brown trout are carnivorous and consume a variety of terrestrial and aquatic vertebrates and invertebrates. When they are small, brown trout eat predominantly aquatic and terrestrial invertebrates, such as zooplankton, insect larvae, crustaceans, molluscs, and terrestrial drift (Scott and Crossman 1973; Hubert et al. 1993; Gislason and Steingrimsson 2004). As they grow, vertebrates especially other fish, begin to figure prominently in the diet. Fish are usually not a large part of the diet until brown trout exceed about 30 cm (Scott and Crossman 1973), although significant piscivory may commence well before this length in some populations (L'Abee-Lund et al. 2002). Brown trout are well known as efficient piscivores, but piscivory is not high in all populations (Saksgård and Hesthagen 2004). In the marine environment, juveniles and adults have been observed to have a high feeding intensity and to be almost exclusively piscivorous (Rikardsen and Amundsen 2005), although crustaceans, surface insects and polychaetes may also be significant in the diet (Knutsen et al. 2001).

6.9.4 Life History

Reproduction.— Brown trout are fall spawners, spawning in late autumn or early winter (Scott and Crossman 1973). In BC, they may spawn as late as early January (Wooding 1994). Spawning is typical of other trout species, and habitat requirements are similar to those of brook trout. Brown trout usually select shallow, gravelly, headwaters, but some individuals make use of rocky shoal areas of lakes (Scott and Crossman 1973). Females excavate shallow redds in gravel and spawn eggs alongside males that release sperm at the same time. The female covers the redd after spawning, and the eggs are left to incubate in the gravel. Brown trout adults often survive to repeat spawn the following year (Mills 1971 cited in Raleigh et al. 1986).

Age at First Reproduction.— Brown trout typically mature at two to four years of age (e.g., Taube 1976; Avery and Hine 1985; Hesthagen et al. 1997; Alp et al. 2003). Some populations mature at substantially older age (Hegge et al. 1991), and occasionally males in some populations mature at one year (Nicola and Almodovar 2002; Hesthagen et al. 2004).

Life Span.— The maximum reported age for brown trout is 38 years (FishBase 2005). Typical maximum age is nearer 10 to 12 years (Hegge et al. 1991; Hesthagen et al. 2004).

Body Size.— Brown trout grow to a large size, particularly the anadromous form. The record is 17.8 kg and 103 cm from Scotland; the Canadian record (from Newfoundland) is 12.9 kg (Scott

and Crossman 1973). Average adult size on inland Canadian streams is roughly 0.5 to 0.9 kg (Scott and Crossman 1973).

Fecundity.— Fecundity is proportional to body size in brown trout and is typically salmonid-like in that eggs are relatively large and fecundity fairly low. Scott and Crossman (1973) note females aged 5 and 6 in the Cowichan River deposited an average of 2020 eggs. Taube (1976) measured fecundity in brown trout from the Platte River, Michigan, and presents numbers in the low hundreds for young fish, up to an average of 1601 for age 5 fish.

6.9.5 Human Interest

Brown trout are a commonly targeted sportfish, and were originally introduced to Canada to enhance angling opportunities. Ironically, introductions were opposed by anglers, who found them too difficult to catch (Scott and Crossman 1973). Biologists were also concerned about the effects of brown trout on native species (Scott and Crossman 1973). Brown trout are generally held in high esteem as a trophy fish, particularly the larger fish, and they continue to be stocked in many jurisdictions. Along with rainbow trout, brown trout are the most widely introduced salmonid globally (Wooding 1994).

6.9.6 Distribution

Native Distribution.— Brown trout are native to Europe and western Asia. They occur north to Iceland and Scandinavia, south to Mediterranean Europe, and east as far as the Aral Sea in Afghanistan (Scott and Crossman 1973).

Present Distribution.— Brown trout have been widely introduced beyond their native range, particularly to North and South America, New Zealand and Australia, and southern Africa (Scott and Crossman 1973; FishBase 2005). They were first introduced to North America in 1883, when they were brought to Michigan and New York (Scott and Crossman 1973; Fuller 2005h). The following year they were introduced to Newfoundland, and there have been successful introductions to all Canadian provinces except for Prince Edward Island, Manitoba and the northern Territories (Scott and Crossman 1973). Brown trout have been stocked into all US states, although most require periodic stocking to maintain population levels (Fuller 2005h).

BC Distribution.— Figure 20 shows the occurrences, as well as stocking records, of brown trout in British Columbia, and the following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. Note that since some waterbodies span more than one MoE region, the total number of unique waterbodies in the province may be different than the sum of counts within each region. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

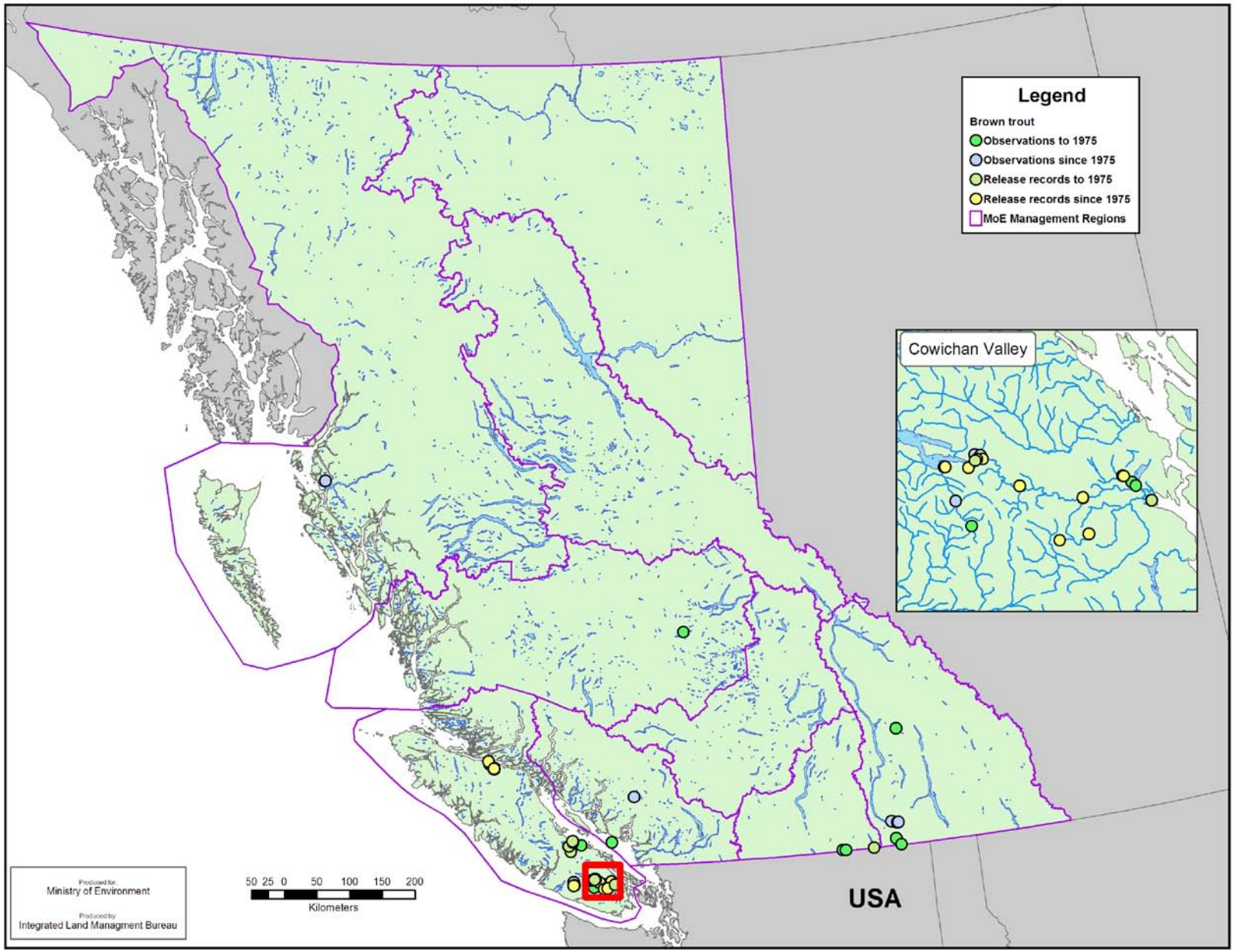


Figure 20. Brown trout occurrences in British Columbia, based on the MoE occurrences database.

Region	No. of waterbodies
1	17
2	2
3	-
4	3
5	1
6	1
7	-
8	2
9	-

Brown trout are not native to British Columbia. Occurrences come primarily from Vancouver Island, with occasional observations from the Lower Mainland, Kootenay, Cariboo and Okanagan regions.

Distribution Trends.— Data indicate an increasing trend in the number of waterbodies in BC with brown trout, but a relatively low number of waterbodies in total. A discussion of the database is presented in Section 6.1.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
3	7	15	25

Stocking Records for BC.— Brown trout have not been widely stocked in BC, based on records in the releases database. All but one stocking event have occurred on Vancouver Island, and of these the vast majority have been in the Cowichan watershed.

Region	No. of waterbodies	No. of releases	Years
1	17	96	1932-1996
2	-	-	-
3	-	-	-
4	-	-	-
5	-	-	-
6	-	-	-
7	-	-	-
8	1	1	1959
9	-	-	-

Mode of Invasion.— Brown trout were intentionally stocked in BC and elsewhere primarily as a sportfish. Such stocking continues to occur elsewhere, but is less common than for other species like rainbow trout. In BC, the last stocking event occurred in 1996. All observations since this time are assumed to be either previously established populations or fish originating from

stockings at or prior to this time. Fuller (2005h) notes that populations in many US states can be maintained only by occasional stocking, although there are clearly many populations in North America and elsewhere that are self-supporting. Brown trout form anadromous populations in many places where conditions permit, and some jurisdictions have seen the spread of brown trout to other watersheds through the ocean (e.g., McDowall et al. 2001). Why this occurs in some locations and not others is not clear, but there is presumably some potential for this mode of invasion in BC. In addition to spread through the sea, brown trout are known to be aggressive competitors (Townsend 1996) and commonly displace native trout species such as brook trout and cutthroat trout as they spread through watersheds (Behnke 1992; Fuller 2005). Brown trout have been occasionally introduced as a biological control agent to control other non-native fish species (Teuscher and Luecke 1996).

An historic article discusses repeated efforts and limited success in establishing a brown trout population in the Cowichan River (Neave 1949). Specifically, Neave (1949) stated that repeated stocking efforts during the 1930s resulted in the establishment of only small numbers in a restricted portion of the river. However, most recent counts of adult brown trout reported several hundred fish residing through much of the Cowichan River from Cowichan Lake to the estuary (n = 892 adults in July 2005, S. Rimmer, pers. comm.); obviously this species has been successful in establishing a relatively large population here over time. The Cowichan River population is considered a resident population (i.e., not anadromous). Yet, the occasional observation of brown trout in surrounding watersheds that have not been stocked (e.g., Caycuse River) suggests some potential for movement through the sea.

In BC no potential impacts on native fish have been documented. Furthermore, there is only limited evidence suggesting the ability of this species to spread to nearby systems. It appears that native species and brown trout successfully co-exist in the Cowichan River, but it is almost impossible to assess impacts to native fish since it was first introduced there.

6.9.7 Threats

Threats to local biodiversity from non-native brown trout are considered significant, to the point that this species has been nominated as one of the 100 worst invasive species, one of only eight fish species on the list (Invasive Species Specialist Group 2005). The brown trout is blamed for reducing native fish populations through predation, displacement, food competition, and hybridization.

Brown trout have been a highly successful invader and have spread beyond their original point of introduction in a number of countries (e.g., Ayllon et al. 2004; Flecker and Townsend 1994; McDowall et al. 2001; Penczak 1999; Waters 1983). Spread to new habitats has occurred through the ocean as anadromous fish stray from their natal streams to invade unoccupied watersheds nearby (Ayllon et al. 2004; McDowall et al. 2001); spread also occurs as fish disperse within a watershed to other unoccupied streams (Waters 1983). In some cases the spread is rapid and expected to be virtually complete to all habitable streams (Ayllon et al. 2004; McDowall et al. 2001). In contrast, spread of brown trout in BC seems to be almost entirely due to authorized stocking programs.

The ecological effects of introduction have been best-studied in New Zealand, where brown trout are now widespread. In these waters they have had a substantial impact on native galaxid fishes, both through competition and predation (Flecker and Townsend 1994; McDowall 2003; McIntosh et al. 1994; Townsend 1996). Brown trout are often highly piscivorous and have the capacity to influence the abundance of native fishes directly through predation (Museth et al. 2003; Penczak 1999). There is abundant evidence that brown trout have replaced indigenous galaxid fishes in many streams (Flecker and Townsend 1994; McDowall 2003). Even though the spread of brown trout has been better studied here than elsewhere, the mechanisms of replacement are not always clear and it is possible that habitat alteration, in addition to direct effects of brown trout, has also played a substantial role (McDowall 2003). Replacement of brook trout populations has been common in North America, even in many locations where brook trout have been introduced (Waters 1983). The precise mechanisms of replacement are unclear here also, but predation, competition, and habitat changes are all implicated. The introduction of brown trout has been implicated in the decline of other salmonids in the US including Dolly Varden in the McCloud River, golden trout in Kern River, populations of cutthroat trout in a number of large rivers, as well as in the demise of the endangered Modoc sucker (*Catastomus microps*) in Rush Creek (Fuller et al. 1999 and references cited therein).

More subtle, community-wide impacts have been reported for benthic invertebrate communities (Flecker and Townsend 1994), a lesson that impacts can occur not just to the more visible components of ecological communities like fish. Insect abundance and density were strongly affected in experimental units with trout, though the effect on invertebrate diversity was relatively minor (Flecker and Townsend 1994). Effects cascaded to lower trophic levels, as algal abundance increased due to lower grazing rates from invertebrates (Flecker and Townsend 1994).

Brown trout are known to hybridize with Atlantic salmon, their congener, sometimes at a high rate (Ayllon et al. 2004). Introgression is minimal or non-existent, as F2 and backcrosses have low viability. Hybridization is unlikely to be a significant issue with native salmonids in BC.

6.9.8 Potential Management Actions

It is generally difficult to eradicate fish species once they become established in the wild. Efforts should therefore be made to prevent dispersal of this species into new habitats, including dissemination of general educational information on the ecological dangers of translocating wildlife, and careful consideration of stocking this species in the wild. Section 5 discusses the various methods available for controlling fish species once they are established.

There are some aspects of this species' biology that may make it easier to control than other fish species, including seasonal reproduction, relatively low fecundity, migration to and use of localized spawning locations, and late maturation. These factors may make it easier to locate fish, to isolate them from other species, and to make control efforts more effective than they would be for some other fish species. For example, if a brown trout population uses a relatively small area for spawning, then this area may be easier to treat with a control methodology than the entire rearing area. This may also make it easier to apply control methods so that they do not significantly influence the distribution and abundance of other fish species. Other aspects of brown trout biology, such as late maturation and relatively low fecundity, may serve to make

it more sensitive to "exploitation" through removal methods. It is not known the extent to which BC populations rely on supplementation, though it is likely that cessation of stocking in BC (the last stocking date was 1996) makes some populations more sensitive to exploitation.

In some locations brown trout are held in very high esteem as a sport fish, and there may be conflicting objectives between conservation of native diversity and management of recreational angling. This may make it difficult to undertake control measures. Some effort may be required to determine the precedence of these objectives and whether they can reasonably coexist (e.g., in the Cowichan River where the brown trout sport fishery is very popular). Indirect risks may arise if it becomes difficult for the public to differentiate between different management actions in different circumstances. For example, it may be difficult to educate the angling community that this is a "bad" species and should not be translocated or protected, particularly if managers are actively releasing other non-native species elsewhere (e.g., brook trout).

Use of fishing regulations to favour selective harvest of brown trout may be more effective than for brook trout. For example, models (Paul 2000 cited in Dunham et al. 2000; Paul and Post 2001) examining the effects of angler harvest show that even catch-and-release fishing with a 10% hooking mortality rate could lead to the extirpation of cutthroat trout in 5 years, whereas brook trout could sustain harvest with no bag limit indefinitely. It may be possible to use this method of control for brown trout. Since many anglers have difficulty properly identifying fish species (Schmetterling and Long 1999), one would need to consider this aspect as part of a change in regulation.

Managers should also consider that no action may be appropriate in cases where there is no threat to non-native species and low likelihood of further spread. In this, and in all cases, it is required to have a good scientific basis for management decisions. For example, an understanding of the mechanisms of invasion is required to make a defensible decision because it is necessary to distinguish between species replacement and species displacement for species invasions. Direct or indirect control of brown trout may be warranted if there is evidence of species displacement. On the other hand, if this species is merely replacing native species that are declining for some other reason (e.g., harvest, changes in habitat quality, etc.) then effort may be better expended addressing that issue.

6.10 Common Carp (*Cyprinus carpio*)[4]



6.10.1 Taxonomy

Family: Cyprinidae (Minnows or carps)
Genus: Cyprinus
Species: carpio
Authority: Linnaeus, 1758

Note: There are several domesticated strains of common carp, including "mirror," "leather," and "koi," often distinguished by colour and scale patterns. All have become established in the wild in some locations (e.g., Barnham 1998; Champion et al. 2002; Dean 2003). There are also different strains of "standard" common carp, which appear to behave differently when released into new environments, some being more invasive (Barnham 1998). Common carp will hybridize with goldfish (*Carassius auratus*), further confusing taxonomic identification in some cases (Barnham 1998).

6.10.2 Habitat

Optimal habitat for carp is generally considered to be low-altitude, still or slow-flowing waters, with access to shallow, vegetated habitat for spawning (McCrimmon 1968). Adhesive eggs are scattered on vegetation (Scott and Crossman 1973). There is no mention in the literature of differential habitat use by juveniles and adults.

Due to their benthic foraging habits adult carp are often associated with fine substrates, such as silt. Carp are tolerant of turbidity and have often been found in waters that are turbid, with poor water quality (McCrimmon 1968). However, they can be found in clear or turbid waters and over most substrate types (Gulf States Marine Fisheries Commission 2003). Trautman (1981) found common carp to be most abundant in Ohio streams enriched with sewage or agricultural run-off, but rare or absent in clear, cold waters and high-gradient streams. At the watershed scale, some of the strongest associations are between carp and human impacts. Carp are common occupants of reservoirs and human-made ponds worldwide; carp also do well in

[4] Image used with permission. Copyright Len Lovshin, Department of Fisheries and Allied Aquacultures, Auburn University, Alabama, USA 36849.

many regulated rivers. Although usually found in freshwater, they have been observed in brackish waters in Atlantic Canada (Scott and Crossman 1973).

6.10.3 Diet

Carp are omnivorous and consume a variety of plants and animals, predominantly benthos.

Juvenile Diets.— Newly hatched carp feed on zooplankton such as rotifers, copepods, and algae, and switch to benthic macroinvertebrates as they grow (McCrimmon 1968). Young of year carp feed on a variety of benthic invertebrates including chironomids, caddis flies, molluscs, ostracods, and crustaceans (McCrimmon 1968).

Adult Diets.— Adult carp are known to eat a wide variety of organisms including insects, crustaceans, annelids, molluscs, fish eggs, fish remains, and plant material (McCrimmon 1968; Scott and Crossman 1973). Though benthos is preferred, carp will at times feed at the surface on floating items (Scott and Crossman 1973). During feeding, carp may uproot and destroy macrophytes.

6.10.4 Life History

Reproduction.— Carp spawn in spring and early summer, when temperatures exceed 17 ˚C (Balon 1995); the spawning season may continue for several weeks (Scott and Crossman 1973). Common carp are broadcast spawners. Females scatter adhesive eggs in the shallows on vegetation or woody debris; they are often accompanied by two or three males during spawning (Scott and Crossman 1973). Eggs typically hatch in less than five days.

Age at First Reproduction.— Over portions of its native range, carp may be sexually mature within one year (Kuliyev and Agayarova 1984 cited in Gulf States Marine Fisheries Commission 2003). However, McCrimmon (1968) indicted sexually maturity normally occurs for males at 3 to 4 years, and for females at 4 to 5 years.

Life Span.— The normal lifespan of carp in North America is usually less than 20 years, though individuals in captivity may live substantially longer (McCrimmon 1968).

Body Size.— Carp often grow 30 to 60 cm in length and weigh 0.5 to 4 kg (Tomelleri and Eberle 1990 cited in Chumchal 2002). However, it is not uncommon for common carp to exceed 20 kg in the wild (McCrimmon 1968).

Fecundity.— Common carp are notably fecund. A typical female (about 45 cm) may produce 300,000 eggs, with some estimates as high as one million over the breeding season (Chumchal 2002).

6.10.5 Human Interest

Carp is a native of Asia, but has been extensively introduced and is now the world's most widely distributed freshwater fish (Barnham 1998). It is the most widely cultured food fish with an annual yield of 200,000 tonnes (Barnham 1998). Carp has a long history of domestication: it

has been propagated in China since the first century and in Germany since the Middle Ages (Barnham 1998). It is one of the most important commercial fisheries in southern and eastern Europe, and is also extensively harvested in many Asian countries (Barnham 1998).

In North America, common carp are not well-regarded as a food fish, although Pflieger (1997 cited in Nico and Maynard 2005) reported that the total weight and value of common carp taken by commercial fishermen in Missouri exceeded that of any other fish. Common carp are not typically targeted by anglers, but can be caught by still-fishing methods with bait (Wooding 1994). In some jurisdictions (including locations within Canada) carp are cultured for use as live bait for anglers (Litvak and Mandrak 1993).

6.10.6 Distribution

Native Distribution.— The common carp has been introduced so widely over so many centuries that its precise origins are uncertain, though its likely native range includes the Black, Caspian and Aral Sea drainages, east into Siberia and China and west as far as the Danube River (Balon 1995).

Present Distribution.— The common carp is now the most widely-distributed freshwater fish in the world (Barnham 1998). It has been introduced to over 140 countries (ISSG 2005), and is found on all continents except Antarctica (Barnham 1998). In the US, common carp are established in all states except for Alaska and Maine (Nico and Maynard 2005). In Canada, they are widely-distributed in Quebec, Ontario, Manitoba, Saskatchewan and British Columbia (Scott and Crossman 1973). Scott and Crossman (1973) list carp as absent from the Maritimes and Alberta, but this may no longer be true. A population in Split Lake, Manitoba (56° 10' N, 96° 0' W) is noted by Scott and Crossman (1973) as the most northerly population in North America. In adjacent Washington State, common carp are found throughout the Columbia and Snake rivers, and the lower reaches of their tributaries (Wydoski and Whitney 2003). They also occur in many lakes, ponds and wetlands throughout Washington (Wydoski and Whitney 2003).

BC Distribution.— The following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. Note that since some waterbodies span more than one MoE region, the total number of unique waterbodies in the province may be different than the sum of counts within each region. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Common carp are concentrated in southern BC and are most common in the Okanagan and Lower Mainland MoE Regions. Carp are established in the Okanagan and Fraser Rivers; these are believed to have derived from natural dispersal from populations in Washington State (Scott and Crossman 1973). The database tracks occurrences of common carp as well as a second category of "carp – general," for observations that do not or cannot distinguish between the common carp and goldfish. At present there is only one record under "carp – general," and this is from Vancouver Island.

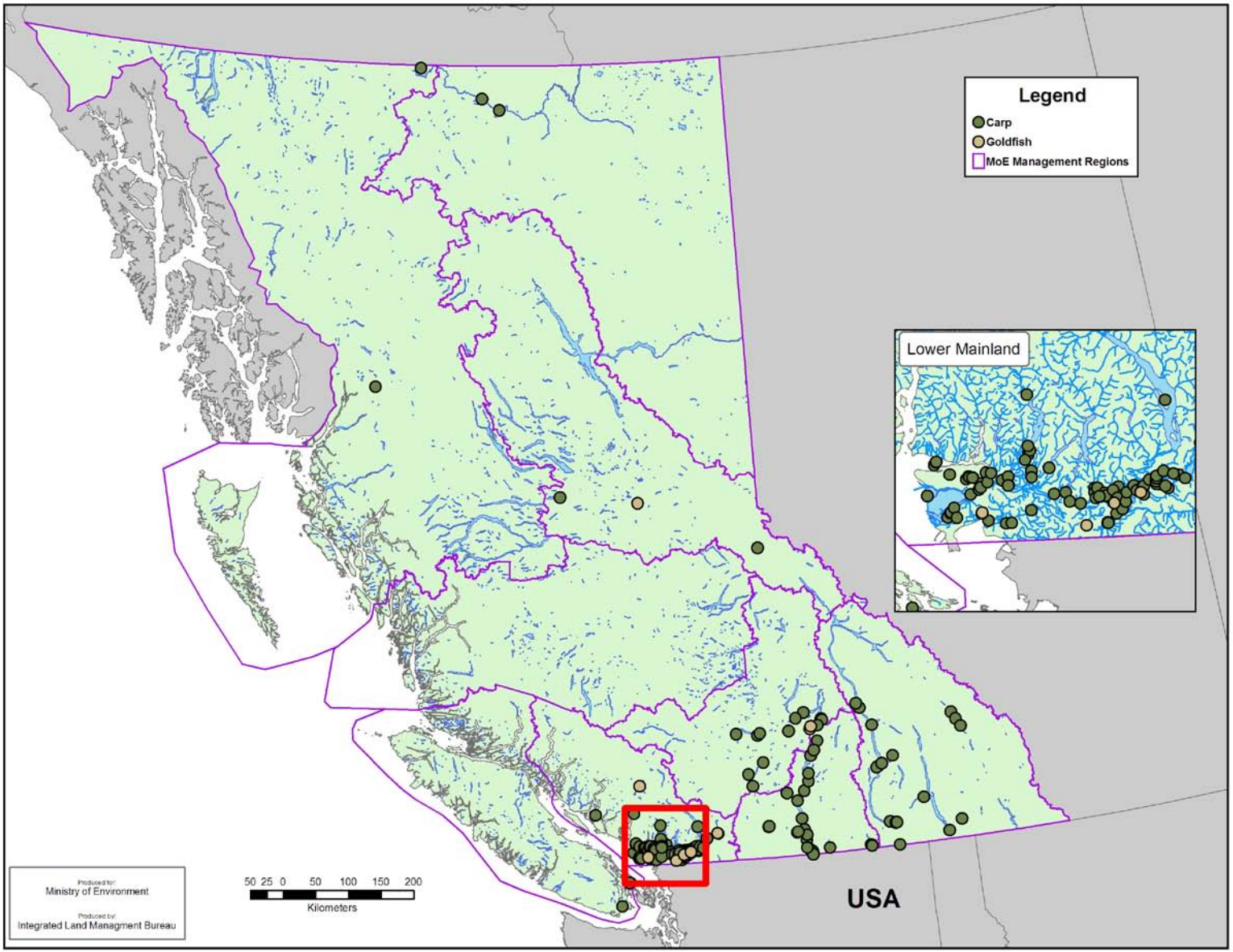


Figure 21. Carp and goldfish occurrences in British Columbia, based on the MoE occurrences database. Note that some specimens were not identified to species, and are tracked simply as “carp / goldfish.”

Region	No. of waterbodies with "carp"
1	2
2	52
3	7
4	6
5	-
6	2
7	2
8	14
9	2

Distribution Trends.— Although many of the observations are not dated, there appears to be a trend of steady spread of this species in BC.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
5	22	53	83

Stocking Records for BC.— There are no records of authorized stocking of common carp in British Columbia.

Mode of Invasion.— Common carp have been intentionally stocked (mostly for food) in the US and to a lesser extent in Canada. They have generally established in waters where introduced (Nico and Maynard 2005). Although typically found in calm waters, common carp are good swimmers that can leap obstacles up to 1 meter high and negotiate torrential flows (FishBase 2005). Distribution has therefore likely been enhanced through both downstream and upstream dispersal. In the Okanagan and Fraser Rivers, carp are believed to have derived from natural dispersal from populations in Washington State (Scott and Crossman 1973).

The extent to which the species is still being introduced in BC is not known, but is presumably not great due to the species' limited appeal (at least within Canada) as a sportfish. Perhaps the greatest risk of introduction comes from the potential release of domesticated pond fish, such as koi. It should be noted that in British Columbia, it is illegal to import koi under the Prohibited Import Live Fish list without a permit. However neither importation nor trade is regulated. In addition, goldfish are not included on this list, and koi could easily be identified as goldfish.

6.10.7 Threats

The IUCN has ranked common carp as one of the 100 worst invasive species because of its widespread distribution and its negative effects on water quality and native ecosystems. The common carp is one of only eight fish species on this list (ISSG 2005). Carp have been introduced as a food and ornamental fish into temperate freshwaters throughout the world. It is considered a pest because of its abundance and its tendency to reduce water clarity and destroy and uproot aquatic vegetation, used as habitat by a variety of aquatic species. Despite

its widespread distribution, the common carp is not well-studied in the wild, either within its native or introduced ranges.

Common carp are considered undesirable due to their adverse effects on fish and wildlife species (Wydoski and Whitney 2003). The most commonly cited effect of carp is the increase in turbidity that is often associated with its presence. This effect has been well-documented in a number of natural and experimental systems (e.g., Richardson et al. 1995; Lougheed et al. 1998; Zambrano et al. 2001; Angeler et al. 2003; Parkos et al. 2003). Benthivorous fish increase the nutrient availability to phytoplankton by transferring sediment-bound nutrients to the water column during bottom foraging (Meijer et al. 1990). The resulting high phytoplankton biomass contributes to high turbidity, which in turn constrains submerged macrophytes. Benthivores also contribute to high non-algal turbidity via resuspension of sediment particles, and by causing mechanical damage to submerged macrophytes (Crivelli 1983; Brabrand et al. 1990). Other factors such as wind fetch, soil type and vegetation cover also have significant influence over turbidity levels (Lougheed et al. 1998) and it is not at all guaranteed that removal of carp will increase water clarity substantially (Lougheed et al. 1998; Angeler et al. 2003). High turbidity can negatively affect visual predators, and can negatively alter water quality for drinking, agriculture and aesthetics.

Other impacts include loss of breeding habitat for both fish and birds, loss of food sources for herbivorous fish and waterfowl and predation on eggs of various fish species including white sturgeon (*Acipenser transmontanus*) in the Columbia River (Fuller et al. 1999 and references cited therein). It may be responsible for the decline of the razorback sucker (*Xyrauchen texanus*) in the Colorado watershed due to predation on eggs, and the Sacramento perch (*Catostomus occidentalis*) due to degradation of spawning grounds (Fuller et al. 1999 and references cited therein).

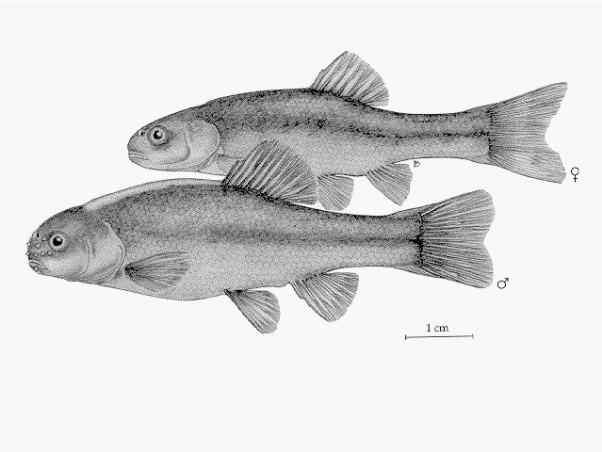
Broader ecological damage attributable to carp includes decreased macrophyte biomass and diversity, increased water nutrient loads, increased algal concentrations, reduced native fish numbers and diversity, decreased macro-invertebrate numbers and diversity, and erosion of stream banks (King 1995 cited in Bomford and Tilzey 1997; Manchester and Bullock 2000). Many of these effects are difficult to separate from the effects of other environmental problems such as river regulation, cattle grazing and riparian vegetation removal. Given its high abundance in some systems (>90% of total fish biomass in some Australian waterbodies) carp are believed capable of considerable ecological detriment such as displacement of native fish (Roberts and Tilzey 1997).

6.10.8 Potential Management Actions

Common carp are difficult to eradicate once established, as evidenced by the considerable and largely unsuccessful effort to do so in Australia and New Zealand (e.g., Roberts and Tilzey 1997; Barnham 1998; Koehn 2004). Poisons have proved effective at killing carp in small impoundments, but have failed to be effective in larger waterbodies (McDowell 1997; Sanger and Koehn 1997). Likewise, increased commercial and recreational angling has not been successful in limiting carp numbers when used as a control technique, and there are fears that this will increase the value of carp and thereby encourage additional spread of the species rather than control local numbers (McDowell 1997; Thresher 1997).

Given the difficulties with control and eradication of non-native carp, efforts are well-spent on trying to prevent human-aided dispersal of this species into new habitats. Information on this species should be included in material used to educate the public on the ecological dangers of translocating wildlife. In particular, there should be some efforts to educate the public on the ecological consequences of releasing domesticated varieties of carp such as koi. Authorities may wish to also evaluate the costs and benefits of regulating the culture and sale of different varieties of carp in BC.

6.11 Fathead Minnow (*Pimephales promelas*)[5]



6.11.1 Taxonomy

Family: Cyprinidae (Minnows or carps)
Genus: Pimephales
Species: promelas
Authority: Rafinesque, 1820

6.11.2 Habitat

Habitat use and preferences vary throughout the Canadian range, but in general fathead minnows prefer still waters (Scott and Crossman 1973) and are found in cool to warm waters. As demersal fish they are usually associated with muddy bottom substrates (Scott and Crossman 1973).

6.11.3 Diet

The feeding habits of fathead minnow have received scant attention, and according to Scott and Crossman (1973) most authors refer to studies by Coyle (1930) when indicating that the species is primarily an algae eater. The digestive morphology (elongated intestine) of fathead is consistent with this diet (Scott and Crossman 1973). The diet of fathead has also been shown to contain detritus, bottom mud, insect larvae and zooplankton (Scott and Crossman 1973).

6.11.4 Life History

Reproduction.— Fathead are spring spawners, and spawning activity commences when water temperatures reach 15.6 ˚C (Scott and Crossman 1973). The spawning season is long and may continue into August (Scott and Crossman 1973).

[5] Image used with permission. Copyright D. McPhail.

Males select spawning sites in shallow water, less than 3 m in depth, on the under-surface of logs, branches, rocks or other cover (Scott and Crossman 1973). After successful courtship, females deposit adhesive eggs on the under-surface, using an ovipositor. Males then fertilize and guard the eggs (Scott and Crossman 1973). Males attempt to entice more than one female to deposit eggs in the nest, and females will usually deposit eggs in more than one nest (Scott and Crossman 1973). No mention is made in the literature of parental care beyond the hatching stage.

Age at First Reproduction.— Growth in some instances is sufficient to allow early hatched fish to become reproductive by the end of that spawning season, though this is likely rare in northern populations (Scott and Crossman 1973). Given the generally short lifespan of fathead minnows, it seems likely that most individuals are reproductive by spring of their first year, though specific details on reproductive ages were not found in the literature.

Life Span.— The fathead minnow is short-lived, rarely living beyond 2 years (Scott and Crossman 1973).

Body Size.— Fathead are small fish usually ranging from 50 to 70 mm as adults, with an average length of 51 mm (Scott and Crossman 1973). Maximum size is reported as 100 mm (FishBase 2005).

Fecundity.— Gale and Buynak (1982) report on a number of aspects of female reproduction based on observations on five pairs of fathead minnows in experimental ponds from May 20 through October 20, 1980. Females produced between 16 and 26 clutches of eggs with the total number of eggs spawned ranging from 6,803 to 10,164 (mean 8,604). Clutch size ranged from 391 to 480 with interspawning intervals ranging from 2 to 16 days (mean 3.9).

6.11.5 Human Interest

The fathead minnow is a common baitfish and laboratory test animal, and a decorative breed (red roseys) is used as an aquarium fish. It is cultured extensively in the US in ponds for use as a baitfish or forage fish for other species such as smallmouth bass (Scott and Crossman 1973; Ludwig and Leitch 1996; Nico and Fuller 2005a).

6.11.6 Distribution

Native Distribution.— The fathead minnow naturally occurs throughout most of central North America, from the southern Great Slave Lake drainage to Chihuahua, Mexico (Scott and Crossman 1973). In Canada, this species is found from the Saint John River in New Brunswick, west through southern Quebec, all of Ontario, and most of Manitoba, Saskatchewan and Alberta. It is naturally absent from British Columbia, although it has been recorded in the Peace River immediately downstream of the BC-Alberta border (Scott and Crossman 1973).

Present Distribution.— The fathead minnow currently exists in almost all states within the US, and Puerto Rico (Nico and Fuller 2005a). It is not present in Alaska or Hawaii (Nico and Fuller 2005a). The species is listed as introduced to several countries in Europe, and also the Middle East (FishBase 2005). There are numerous populations of fathead minnow in adjacent

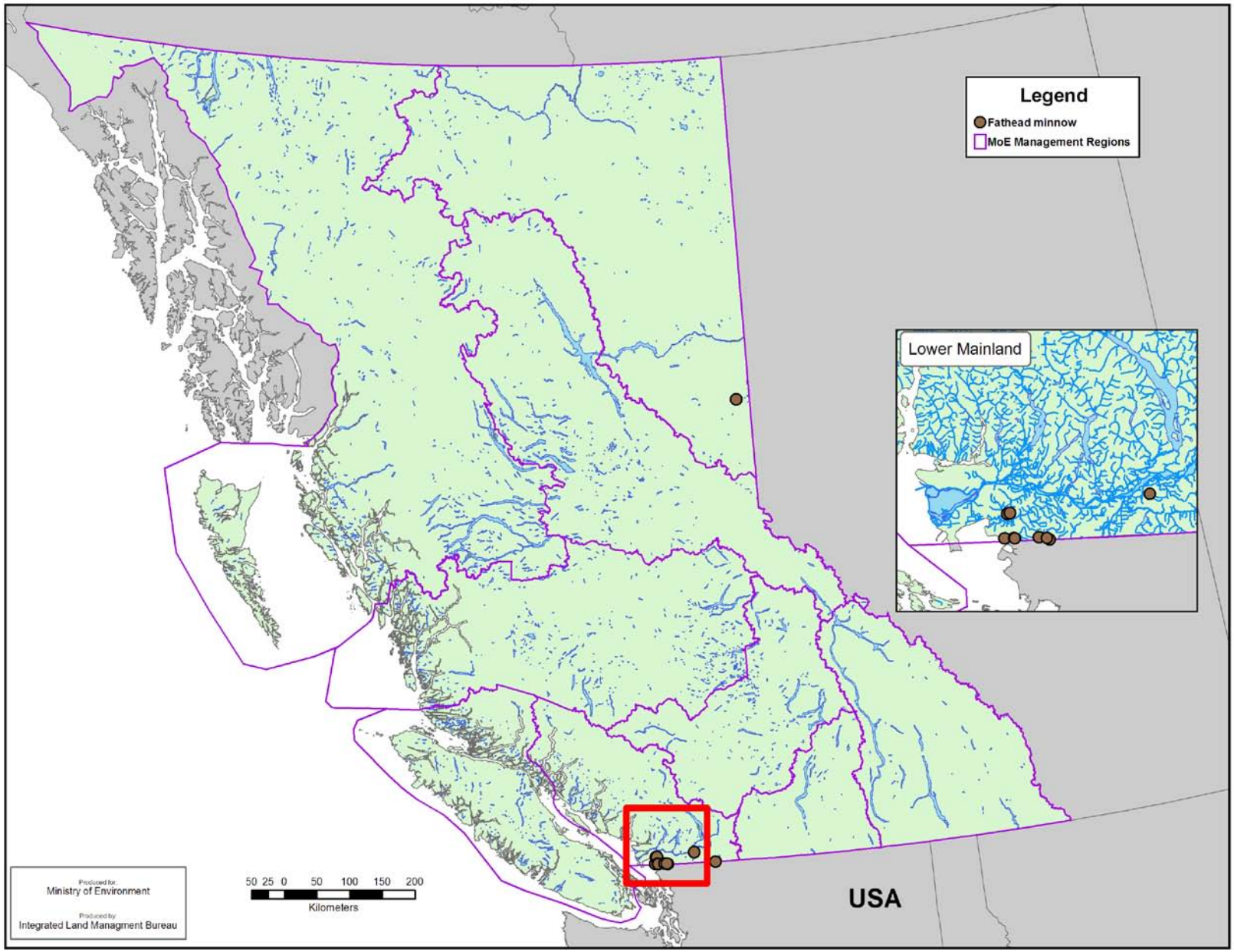


Figure 22. Fathead minnow occurrences in British Columbia, based on the MoE occurrences database.

Washington State, including waterbodies where they are raised as a baitfish (Wydoski and Whitney 2003).

BC Distribution.— The following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Region	No. of waterbodies
1	-
2	10
3	-
4	-
5	-
6	-
7	-
8	-
9	1

The fathead minnow is not native to BC waters. The species has been occasionally observed in the Lower Mainland, primarily in the lower Fraser Valley, and a single record comes from the Peace region.

Distribution Trends.— The majority of records are undated, but dated records suggest introduction of this species to BC has been recent. Although there is an increasing trend in the number of waterbodies in BC with fathead minnow, there are still a relatively low number of waterbodies in total.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
-	-	4	11

Stocking Records for BC.— There are no records of authorized stocking of fathead minnow in British Columbia.

Mode of Invasion.— The fathead minnow is a common baitfish and laboratory test animal, and a decorative breed (red roseys) is used as an aquarium fish. Each of these has been implicated in the introduction of this species to non-native waters. The release of baitfish is believed to be by far the biggest contributor to the spread of fathead minnow in the US (Nico and Fuller 2005a). Fathead minnow has also been widely-introduced (authorized and unauthorized intentional introductions) as a forage fish for sportfish enhancement (Scott and Crossman 1973; Nico and Fuller 2005a) and occasionally for use as a mosquito-control agent (Nico and Fuller 2005). In the US, this species has established populations in most locations where it has been introduced (Fuller et al. 1999). It is unclear how the minnow was introduced in BC: possibly, it was brought in as a baitfish in the past.

6.11.7 Threats

Despite their broad distribution and considerable abundance in non-native waters, the threats from fathead minnow to native fish species have not been quantified or described in any detail, and are largely unknown. In general, there is exceedingly little scientific or even anecdotal evidence of adverse ecological effects from the introduction of this species. This in itself may indicate that impacts to native biodiversity are usually not severe.

Non-native fathead minnow have been implicated as a threat to young of the endangered Colorado pikeminnow (*Ptychocheilus lucius*; Karp and Tyus 1990), and there is a general concern that this and other non-native fishes may compete with or prey on endangered species such as the Lost River sucker (*Deltistes luxatus*) and shortnose sucker (*Chasmistes brevirostris*) in Oregon and northern California (U.S. Fish and Wildlife Service 1993). Non-native fishes, including fathead are likely at least partially responsible for the decline of the Chiracahua leopard frog (*Rana chiricahuensis*) in southeastern Arizona (Rosen et al. 1995). Abundance of fathead minnow in Klamath Lake, Oregon, was inversely correlated with abundance of native minnows and suckers, implying competition with native species (Wydoski and Whitney 2003).

6.11.8 Potential Management Actions

Like virtually all non-native fish species, fathead minnow are difficult to eradicate once established (see Section 5). Efforts are best spent in preventing dispersal of this species into new habitats, including dissemination of general educational information on the ecological dangers of translocating wildlife and use and enforcement of specific regulations.

Use of live bait is currently forbidden in BC; this prohibition limits opportunities for release of this (and other) bait fish species in the province. However, fathead minnows are raised as baitfish elsewhere in North America and this has been one of the primary modes of introduction of this species (Nico and Fuller 2005a; Litvak and Mandrak 1993).

6.12 Goldfish (*Carassius auratus*)



6.12.1 Taxonomy

Family: Cyprinidae (Minnows or carps)
Genus: Carassius
Species: auratus
Authority: Linnaeus, 1758

Note: There is some disagreement over taxonomic status, with some authors claiming that goldfish are a subspecies or domesticated form of Crucian carp (*Carassius carassius*), a species with which goldfish readily hybridize (Nico 2005; Hänfling et al. 2005). There are numerous domesticated forms of goldfish, including many colour and morphologic variants. When released into the wild, the coloured forms generally revert to the wild, olive-green colour morph (Scott and Crossman 1973).

6.12.2 Habitat

Goldfish typically occur in freshwater, but tolerate brackish water of low salinity (Gulf States Marine Fisheries Commission 2003). In North America, goldfish typically occur in warmwater ponds, ditches, and rivers, with stagnant or slow-flowing water, abundant vegetation, and muddy bottom, though they are also found in cooler regulated systems (Gray and Dauble 2001; Gulf States Marine Fisheries Commission 2003; FishBase 2005). In Australia, goldfish are almost solely restricted to highly altered habitats such as agricultural ditches or small reservoirs (Morgan et al. 2004). Goldfish are tolerant of turbidity, pollution, and low oxygen levels, more so than most species (Gulf States Marine Fisheries Commission 2003). Spawning habitat is vegetated shallow areas (Scott and Crossman 1973).

6.12.3 Diet

Goldfish are omnivorous. Young feed on zooplankton; whereas older fish feed on detritus, algae, macrophytes, zooplankton and benthos (Scott and Crossman 1973; Magalhaes 1993; Richardson et al. 1995; Speczian et al. 1997; Gulf States Marine Fisheries Commission 2003; FishBase 2005; Nico 2005). Goldfish are known to put considerable grazing pressure on

macrophytes (Richardson et al. 1995; Gulf States Marine Fisheries Commission 2003) and may be used to control growth of aquatic vegetation (Gulf States Marine Fisheries Commission 2003).

6.12.4 Life History

Reproduction.— Goldfish spawn in late spring and early summer. In the mid-Columbia River, goldfish spawn at temperatures of 10˚ to 16˚ C (Gray and Dauble 2001). Goldfish are broadcast spawners. Females scatter numerous adhesive eggs in the shallows on vegetation or woody debris (Scott and Crossman 1973; FishBase 2005); they may be accompanied by more than one male during spawning (Scott and Crossman 1973). Eggs hatch in three to six days, depending on temperature (Scott and Crossman 1973; Gulf States Marine Fisheries Commission 2003). Cold winter water temperature is required for proper ova development (FishBase 2005).

Age at First Reproduction.— Goldfish reach sexual maturity in one to three years (Gulf States Marine Fisheries Commission 2003).

Life Span.— Goldfish typically live six to seven years, but have lived as long as 30 years in captivity (Gulf States Marine Fisheries Commission 2003).

Body Size.— Goldfish are typically 120 to 220 mm in length, but have been grown to 410 mm in captivity (Gulf States Marine Fisheries Commission 2003; FishBase 2005).

Fecundity.— Goldfish have very high fecundity; females have 191,000 to 403,000 eggs (mean 268,000; Gulf States Marine Fisheries Commission 2003).

6.12.5 Human Interest

Although goldfish are widely-distributed they are not an important food or sportfish. They are used extensively as a laboratory animal, and are cultured in large numbers for the aquarium and bait trade. They are a common ornamental fish in ponds and aquaria, and have been selectively bred for various colours and morphologies.

6.12.6 Distribution

Native Distribution.— Notwithstanding some of the recent taxonomic controversy, goldfish originated in China and are native to eastern Asia.

Present Distribution.— Goldfish are widely-distributed. They have been introduced to parts of Asia, Europe, North and South America, Africa and Australia. They are cited as being the first non-indigenous fish species introduced to North America, sometime in the early 17th century (Nico 2005). Goldfish have been collected from all US states except Alaska, and are believed to be established in most (Nico 2005). They have only a sporadic distribution in Canada (Scott and Crossman 1973). Whoriskey and Brown (1994) observed goldfish in streams and ponds in the St. Lawrence River basin, Quebec, and found them to have good overwinter survival; this may indicate there is abundant available habitat for goldfish in many parts of Canada.

BC Distribution.— The following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Region	No. of waterbodies
1	-
2	7
3	2
4	-
5	-
6	-
7	1
8	-
9	-

There are a total of 12 records of goldfish in the MoE database, 10 of which are in unique waterbodies. The majority of observations come from the Lower Mainland Region. The database tracks occurrences of goldfish as well as a second category of "carp – general," for observations that do not or cannot distinguish between the common carp and goldfish. At present there is only one record under "carp – general," and this is from Vancouver Island.

Distribution Trends.— There is some evidence of goldfish spreading in BC, though the number of observations is still very low.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
1	2	9	10

Stocking Records for BC.— There are no records of authorized stocking of goldfish or carp in British Columbia.

Mode of Invasion.— Historically, the primary mode of introduction and spread of goldfish in North America has been intentional release through authorized fish stocking programs or unauthorized release of aquarium stock, pondfish, and bait fish (Scott and Crossman 1973; Gulf States Marine Fisheries Commission 2003; Nico 2005). Goldfish continue to be used as bait fish in some jurisdictions (Gulf States Marine Fisheries Commission 2003; Nico 2005), which has resulted in additional spread of this species (Litvak and Mandrak 1993). Natural dispersal and flooding events have also increased the distribution of goldfish in some North American waters (Burr et al. 1996). This species has been established or at least reported in all US states except Alaska, and 21 of these states have confirmed established populations (Fuller et al. 1999 and references cited therein). A study in Illinois concluded that goldfish only become established in severely disturbed waters (Laird and Page 1996 cited in Fuller et al. 1999).

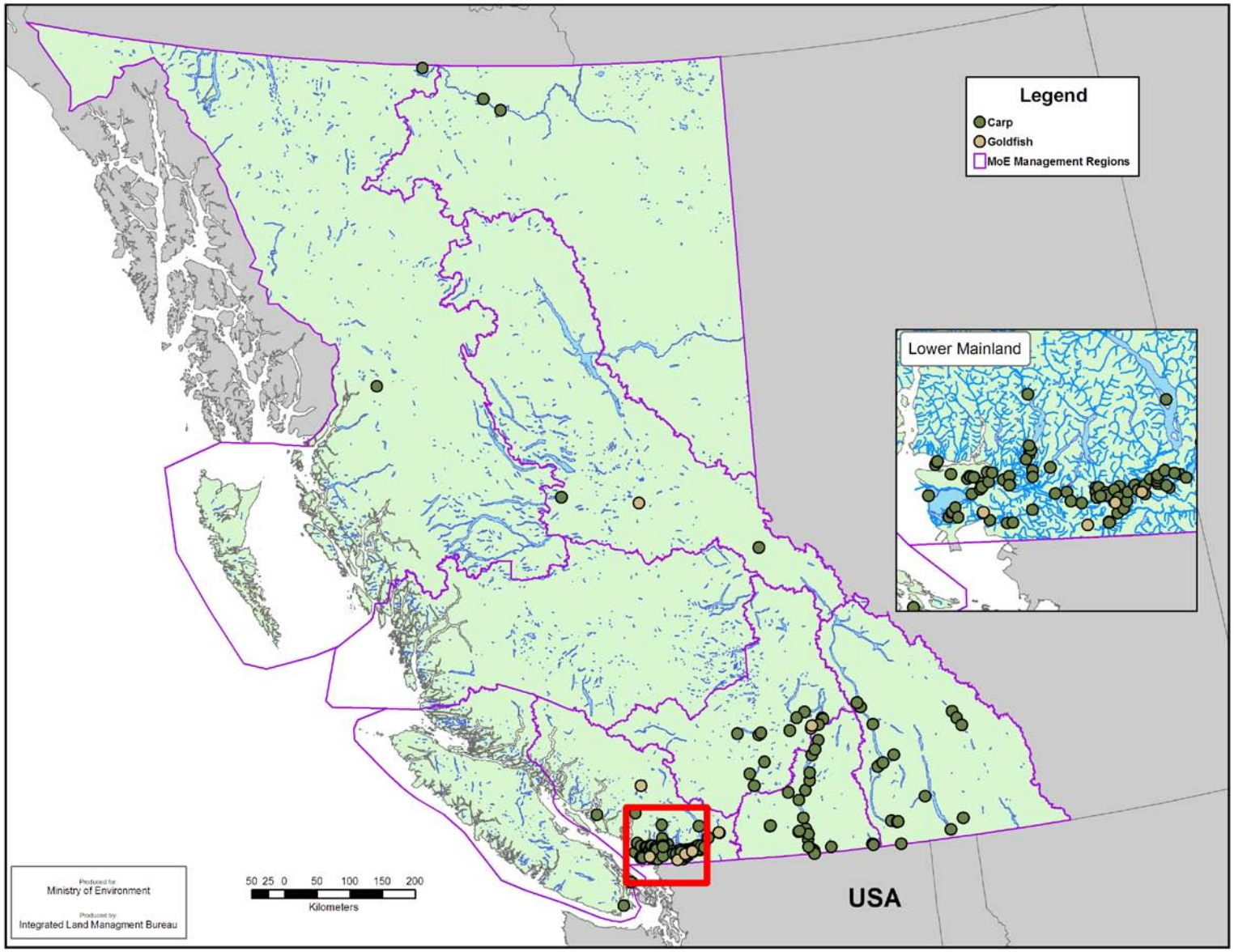


Figure 23. Carp and goldfish occurrences in British Columbia, based on the MoE occurrences database. Note that some specimens were not identified to species, and are tracked simply as “carp / goldfish.”

6.12.7 Threats

Goldfish have been widely-introduced to a variety of habitats throughout the world, over a long period of time. This, along with the fact that negative effects are seldom reported, may be some evidence that impacts to native biodiversity are not severe.

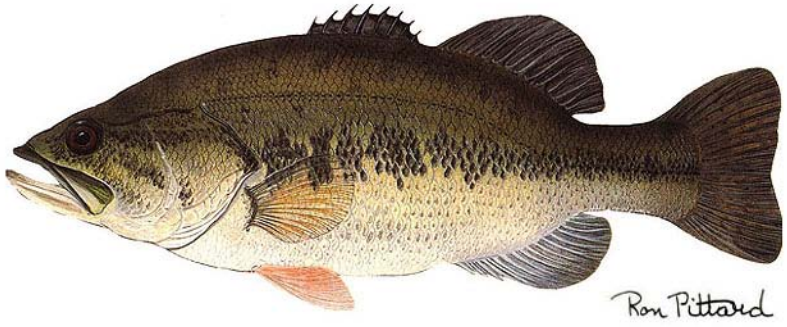
Goldfish are occasionally listed as a pest species, though they often pose relatively minor threats to native fish and wildlife, in part because of the highly-disturbed and altered habitats in which they tend to occur (Morgan et al. 2004). In some cases their status as pest may reflect high abundance rather than ecological effects, per se (Morgan et al. 2004).

The most commonly cited ecological effects of goldfish are generation of turbidity and destruction of macrophytes (Richardson and Whoriskey 1992; Richardson et al. 1995; Gulf States Marine Fisheries Commission 2003; Nico 2005). Richardson et al. hypothesized that the large increase in turbidity may be adaptive for goldfish, as a deterrent to visual predators. Where Crucian and common carp are important species, there are some worries about the effect of hybridizing goldfish on the genetic integrity of wild populations, though this is not a biodiversity concern in BC (Burr et al. 1996; Hänfling et al. 2005). It is likely that goldfish compete with native species for food and space and large numbers can seriously impact native fish habitats. In the US goldfish introductions are believed to be responsible for the initial decline of the now endangered Pahrump poolfish (*Empetrichthys latos latos*) due to competition and predation of young (Fuller et al. 1999 and references cited therein).

6.12.8 Potential Management Actions

Like virtually all non-native fish species, goldfish are difficult to eradicate once established. Jumpoff Joe Lake, in eastern Washington, was treated repeatedly – and unsuccessfully – with rotenone to eradicate goldfish (Wydoski and Whitney 2003). Efforts are likely best-spent on trying to prevent human-aided dispersal of this species into new habitats. Since goldfish are often associated with degraded and altered aquatic habitats, avoiding such impacts may help prevent the general spread of this species. Use of live bait is currently forbidden in BC: this prohibition limits opportunities for release of this (and other) species. However, goldfish are raised as baitfish elsewhere in North America and the use of goldfish as bait has been implicated in the species spread (Litvak and Mandrak 1993). Information on goldfish should be included in material used to educate the public on the ecological dangers of translocating wildlife. In particular, there should be some efforts to educate the public on the ecological consequences of releasing domesticated varieties of fish such as goldfish. Authorities may wish to also evaluate the costs and benefits of regulating the culture and sale of goldfish in BC, particularly as this species is easily confused with koi, a species that is more regulated for trade purposes.

6.13 Largemouth Bass (*Micropterus salmoides*)



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6.13.1 Taxonomy

Family: Centrarchidae (Sunfishes)
Genus: Micropterus
Species: salmoides
Authority: Lacepède, 1802

6.13.2 Habitat

Largemouth bass are primarily warmwater lake-dwelling fish. Their general habitat is small, shallow lakes, shallow bays of larger lakes, and occasionally the still backwater habitats of large, slow rivers (Scott and Crossman 1973). These bass are almost always found in areas of soft sediments, in and around extensive macrophyte growth, at depths less than 6 m (Scott and Crossman 1973). When co-occurring with smallmouth bass, the two rarely share the same microhabitat (Scott and Crossman 1973).

Spawning Habitat.— Males provide all parental care and build nests in littoral areas. Nests are usually constructed in vegetated areas over muddy or marl bottom sediments, though they may more occasionally be found in sandy, gravelly areas (Scott and Crossman 1973). Nesting areas are usually in sheltered, protected bays.

6.13.3 Diet

Juvenile diets.— The diet of largemouth bass changes as it increases in size. Very young fish are planktivorous, feeding on zooplankton, but switch to drift insects and benthic invertebrates as they increase in length (Shrader and Weldon 2004). Fish (including young largemouth bass) enter into the diet when the bass reach a total length of about 50 mm (Scott and Crossman 1973). Growth rates improve markedly when young fish make the switch to piscivory (Olson 1996; Olson and Young 2003).

Adult Diets.— Adult largemouth bass are visual predators that generally target fish, though insects, crayfish, other invertebrates, and amphibians will also be taken depending on the availability of different food items (Scott and Crossman 1973; Godinho et al. 1997).

6.13.4 Life History

Reproduction.— Largemouth bass reproduction is similar to that of smallmouth bass. Males provide all parental care. In the spring, they acquire territories in the littoral region where they build nests and mate (sometimes with many females). Nest building activity usually commences when water temperatures reach about 15.6 ˚C, with spawning starting when temperatures reach a degree or two warmer (Scott and Crossman 1973). Spawning occurs in the late spring to mid-summer, with a peak usually in early to mid-June. Male largemouth bass are aggressive and territorial; they vigorously defend their nests and juveniles within the nest area for up to a month post-hatching (Scott and Crossman 1973).

Age at First Reproduction.— In Canada, males are mature in 3 to 4 years, and females in 4 to 5 years (Scott and Crossman 1973).

Life Span.— Maximum reported age is 23 years (FishBase 2005), though Marchetti et al. (2004a) list maximum life span as 9 to 16 years.

Body Size.— FishBase (2005) lists a maximum size of 97 cm total length and 10.1 kg, however, Scott and Crossman (1973) note that fish of 2.7 to 3.6 kg are prize-winning sizes in Ontario and Quebec fishing derbies. Those taken by anglers in Ontario and Quebec usually do not exceed 0.9 to 1.3 kg (Scott and Crossman 1973).

Fecundity.— Marchetti et al. (2004a) list maximum fecundity as within the range of 10,000 to 100,000 eggs per female. Scott and Crossman (1973) list typical fecundity as 2,000 to 109,314 eggs per female, or 2,000 to 7,000 eggs per pound (.45 kg) of female.

6.13.5 Human Interest

Largemouth bass are held in high esteem by many anglers, and their attractiveness to anglers is often cited as the primary reason for the spread of this species. Largemouth bass were widely stocked throughout North America by government fish and wildlife agencies. The species is most commonly stocked now (illegally) by anglers wishing to enhance sport fishing opportunities (Rahel 2000), although it is still stocked by some agencies in the US. Largemouth bass are present in 85% of lowland warmwater lakes in Washington State, 84% of similar lakes in Oregon, and 74% of similar lakes in northwest California (Bonar et al. 2005). Although they may have been intentionally stocked into only a few BC lakes by authorized means in the early 1900s, there is believed to be an increasing interest by angling groups to introduce largemouth (and smallmouth) bass into suitable habitats in BC (Bonar et al. 2005). Because largemouth bass are tolerant of high temperatures and turbidity, and have a fast growth rate, the species has also been used to stock warmwater farm ponds (Scott and Crossman 1973).

6.13.6 Distribution

Native Distribution.— Largemouth bass have been translocated to so many locations that its precise natural distribution is not known. Its broad distribution is similar to that of smallmouth bass, but it is common only in a smaller, more southern area (Scott and Crossman 1973). It is naturally absent from western North America, the Maritimes, and the New England states (Fuller 2005e). In Canada, smallmouth bass occur naturally in the Great Lakes and St. Lawrence region of southern Quebec and Ontario, plus a small portion of the Red River watershed in Manitoba (Scott and Crossman 1973). From this northern limit, largemouth bass extend naturally through most of eastern central US to northeastern Mexico (Scott and Crossman 1973).

Present Distribution.— The largemouth bass is now widely distributed worldwide (FishBase 2005). In North America, it is present in Mexico, all US states except Alaska, and many locations in southern Canada. It has been widely introduced in South America, Asia, Europe, and Africa (FishBase 2005). Largemouth bass are widely distributed in adjacent Washington State (Wydoski and Whitney 2003).

BC Distribution.— The following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Region	No. of waterbodies with "largemouth bass"	No. of waterbodies with "bass/sunfish - general"	No. of waterbodies with one or both
1	-	5	5
2	36	5	40
3	-	1	1
4	28	7	33
5	-	-	-
6	-	-	-
7	-	-	-
8	7	3	10
9	-	-	-

The database tracks occurrences of smallmouth and largemouth bass separately, but there is also a third category of "bass/sunfish - general," for observations that do not or cannot distinguish between the two bass species and/or sunfish species. We assume that some of these observations are of largemouth bass.

Largemouth bass are not native to British Columbia. Based on the MoE records, populations exist only in the Lower Mainland, Kootenays, and Okanagan Regions. Largemouth bass occurrences appear to have a more restricted distribution than smallmouth bass occurrences.

Distribution Trends.— The following data come from the occurrences database, and indicate an increasing trend in the number of waterbodies with largemouth bass. Most observations have

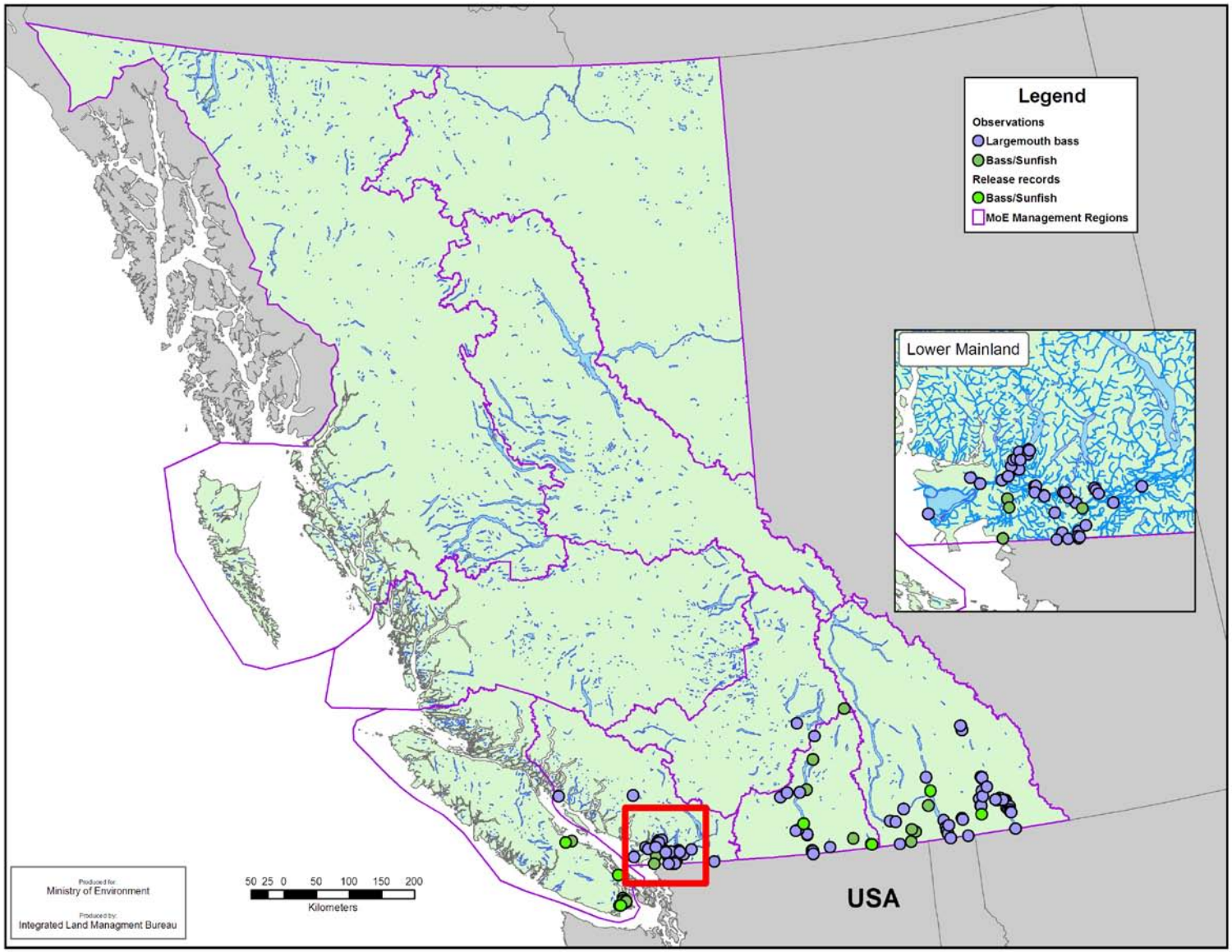


Figure 24. Largemouth bass occurrences in British Columbia, based on the MoE occurrences database. Note that some specimens were not identified to species, and are tracked simply as "bass / sunfish." Also shown on this map are the sites for authorized releases.

been made in the last 30 years. Note that some records in the database are not dated, so the total number of waterbodies exceeds the value in 2005.

	Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
largemouth bass	1	8	41	71
Bass/sunfish	-	1	6	21

Stocking Records for BC.— There are no specific records in the releases database of authorized stocking of largemouth bass in British Columbia. However, there are nine records marked as "bass/sunfish (general)," which date to the early 1900s. These introductions occurred in seven lakes in the Vancouver Island, Kootenay and Okanagan Regions, listed in the following table.

Waterbody	**MoE Region**	**Year**
St. Mary Lake	1	1920
Langford Lake	1	1901
Florence Lake	1	1901
Spider Lake	1	1923
Moyie Lake	4	1901
Fraser Lake	4	1923
Christina Lake	8	1901

It has been assumed that these introductions were of smallmouth bass, but it is possible that stocks were largemouth bass, pumpkinseed sunfish, or a mixture. Of these lakes, there are observations in the occurrences database of largemouth bass in Fraser Lake, Christina Lake, as well as Moyie River. However, it is clear that the great majority of introductions have been unauthorized.

Mode of Invasion.— Largemouth bass have been spread through authorized and unauthorized introductions, usually for the purpose of establishing sport fisheries for this species (Scott and Crossman 1973; FishBase 2005; Fuller 2005e). In BC, the majority (if not all) of occurrences are believed to be the result of illegal introductions and subsequent spread. Largemouth bass are primarily lake-dwelling, but also inhabit some larger, slow-moving, warmwater rivers. Although it is not usually listed as particularly invasive species (i.e., one that spreads aggressively on its own), it is capable of dispersing naturally through low gradient river systems.

6.13.7 Threats

The IUCN has ranked largemouth bass as one of the 100 worst invasive species because of its widespread introduction and its negative effects on native fish species. The largemouth bass is one of only eight fish species on this list (ISSG 2005). In general, there is a more limited literature on the ecological effects of largemouth bass in comparison to that of smallmouth bass. Section 6.16.7 details the numerous ecological effects of non-native smallmouth bass, including local extinction of littoral fishes, substantial food web effects, and population-level effects on

resident and anadromous fishes. Species documented to be affected by bass introductions include a number of chub and dace species in the US. Furthermore, bass introductions are thought to be responsible for the decline of ranid frogs and the tiger salamander in California, as well as the Chiricahua leopard frog in Arizona (Fuller et al. 1999 and references therein). Experience and available research indicates that effects are likely to be similar for both species, since both are top-level piscivores with similar diets and habitat requirements.

Bonar et al. (2005) recently reported predation pressure by non-native largemouth bass in three small Pacific Northwest (US) lakes containing native salmonids. They indicated that 5 to 20% of coho smolt production in these lakes was consumed by juvenile and adult bass, a percentage that most would consider to be substantial. Although landscape-level estimates (i.e., estimates of predation effects across large geographic areas) have not been made, it seems reasonable to assume that given their widespread introductions in the western US (Bonar et al. 2005), largemouth bass have had a considerable impact on native salmonid production, as well as on other species.

Threats Summary.— Largemouth bass, like smallmouth bass, are considered a significant threat to aquatic biodiversity in BC, particularly in southern areas where there is more available habitat. Bass are attractive to anglers and have a history of being actively stocked (both legally and illegally) in BC and elsewhere in North America to "enhance" angling opportunities. Bass are top-level carnivores, the trophic level most likely to cause substantial ecological effects (Moyle and Light 1996a, b; Marchetti et al. 2004a). Given the general topography of BC, the spread of largemouth bass depends primarily on active transport by humans.

6.13.8 Potential Management Actions

Largemouth bass are relatively easy to catch in large numbers using minnow traps, nets, and electrofishing, however, extirpating a population of bass is likely to require the coincident use of several of the methods discussed in Section 5. Indeed in some systems, such as large lakes with existing populations of other desirable fish species it is essentially impossible to extirpate bass once they are established. Efforts are therefore best spent on preventing introductions to habitats that do not currently support bass. These efforts will require regulation, enforcement, education and stewardship. In theory, adequate statutes and regulations already exist: importation of bass is prohibited, as is the possession, transport and introduction of any live fish into natural waters without a permit (see Section 2.2). Unfortunately, these regulations are not stringently enforced, and compliance is insufficient to prevent introductions. Education is most needed for persons likely to translocate these (or other) fish. Strategies should be developed to ascertain who is likely to introduce fish, and to target communication at such groups and individuals. Strategies should also be developed that would aid public stewardship of freshwater bodies in BC, to prevent the introduction and spread of aquatic species in BC. The comparatively limited distribution of largemouth bass in BC suggests the spread of this species may be less problematic than that of smallmouth bass, or that illegal activities have focussed on smallmouth bass. However, if Washington state is used as a reference point, there is clearly considerable potential (and presumably desire from some groups) to have largemouth bass spread in BC.

6.14 Northern Pike (*Esox lucius*)



6.14.1 Taxonomy

Family: Esocidae (Pikes)
Genus: Esox
Species: lucius
Authority: Linnaeus, 1758

6.14.2 Habitat

In Canada, pike habitat is usually clear, warm, still or slow-moving water with abundant vegetation (Scott and Crossman 1973). Northern pike are ambush predators and use vegetation as cover while waiting for prey, although they may use other forms of cover also (Inskip 1982). Most pike inhabit lakes; they are not well-adapted to fast-moving water, although they do make use of slow-moving, backwater riverine habitats (Inskip 1982). Northern pike generally occur in fairly shallow water, and most are caught at less than 5 m depth (Scott and Crossman 1973).

Spawning Habitat.— Northern pike typically spawn in the heavily vegetated floodplains of rivers, marshes, and bays of large lakes (Scott and Crossman 1973). Lakes account for most pike rearing habitat in North America, but it is common for pike to migrate up tributaries for spawning (Inskip 1982). Northern pike broadcast spawn over vegetation and have a weak preference for inundated grasses and sedges (Inskip 1982 and references therein), though do not appear to have preferences for underlying substrates. Scattered vegetative debris will be used when flooded vegetation is absent (Inskip 1982 and references therein). Northern pike do not generally do well in reservoirs with large fluctuations in water level due the absence of littoral vegetation, and pike are known to delay spawning if water levels do not provide sufficient spawning habitat (Inskip 1982).

6.14.3 Diet

Northern pike are omnivorous carnivores that concentrate on vertebrate prey from an early age. Scott and Crossman (1973) note that pike switch from zooplankton and aquatic insects to small fish within only 7-10 days of yolk absorption, or when they are about 2 inches in length. Fish are the primary prey item, but adults will also prey heavily at times on other vertebrates, such

as amphibians, crayfish, mice, muskrats and ducklings (Scott and Crossman 1973). They are said to have voracious appetites.

6.14.4 Life History

Reproduction.— Northern pike do not build nests. They are spring spawners, and spawn usually immediately after ice melt (Scott and Crossman 1973), in April to early May. Males and females pair based on body size, large females are sometimes attended by two males (Scott and Crossman 1973). Eggs and sperm are broadcast widely over shallow, vegetated areas, via repeated spawning bouts (Scott and Crossman 1973). Eggs usually hatch in 12 to 14 days, but can hatch in as little as 4 to 5 days in warmer waters; this is followed by 6 to 10 days of inactivity while the yolk is absorbed (Scott and Crossman 1973).

Age at First Reproduction.— In southern Canada, females mature at 3 to 4 years and males at 2 to 3 years (Scott and Crossman 1973). In the north both sexes mature later, females at 6 years and males at 5 years (Scott and Crossman 1973).

Life Span.— In southern Canada, life span is about 10 to 12 years, whereas in the north it can be as high as 24 to 26 years (Scott and Crossman 1973). Maximum reported age is 30 years (FishBase 2005).

Body Size.— FishBase (2005) lists the maximum body size as 137 cm for males and 150 cm for females; maximum weight is listed as 35 kg. Lengths of near 1m are not uncommon (Scott and Crossman 1973).

Fecundity.— Egg number is proportional to body size and egg numbers have been reported to be as high as 595,000 (Scott and Crossman 1973). Scott and Crossman (1973) list egg number as 9000 per pound (454 g) of female, or 32,000 eggs on average for mature females.

6.14.5 Human Interest

Northern pike are an important sport and commercial fish species. They make up a significant portion of the commercial fishery in many locations (Scott and Crossman 1973). As a sport fish, northern pike are also important in many locations, especially in the winter ice fishery (Scott and Crossman 1973). In some places, northern pike are considered a pest species due to their effect on other target fish species (Scott and Crossman 1973; Fay 2002).

6.14.6 Distribution

Native Distribution.— The northern pike is a freshwater species, with occasional presence in weakly brackish water. It has a circumpolar distribution; it is widespread in northern North America, Europe and Asia. In Canada, northern pike are present throughout most regions, including many parts of the Yukon and Northwest Territories and Nunavut. It is naturally absent west of the Rockies and also in the Maritimes, the Gaspé Peninsula, and northern coast of Labrador (Scott and Crossman 1973). In Alaska, the northern pike is native north and west of the Alaska Range. It is naturally present only in the northern of the lower 48 states, bounded west by the Rockies and east by the Appalachian mountains (Scott and Crossman 1973).

Present Distribution.— Northern pike have not been widely-introduced beyond their native countries. Introductions have been primarily to a few countries in southern Europe and northern Africa (FishBase 2005). However, northern pike have been widely translocated within North America and Europe. Historically, introductions were often authorized; new introductions are primarily unauthorized. There is considerable concern about the spread of pike in southcentral Alaska (Fay 2002). Northern pike were illegally introduced into the Coeur d'Alene system in Idaho in 1972 and were firmly established by the mid-1980s (Wydoski and Whitney 2003). From there they expanded their range into the Spokane River, Washington (Wydoski and Whitney 2003). The species will likely expand its range within the Columbia system over time (Wydoski and Whitney 2003).

BC Distribution.— Northern pike are native in British Columbia only east of the Rockies, in the north and northeast of the province. Figure 25 shows both the assumed native range of northern pike in BC, as well as all individual occurrences in the province. The vast majority of observations are within the native range, with only two observations outside this range, in the Omineca Region. Additional waterbodies may be occupied by northern pike, but these are not recorded in the occurrences database. In September 2005, there was a confirmed report of two northern pike captured in Haha Lake, a lake on Haha Creek, which flows into the northwest of Lake Kookanusa in southeast BC. This represents the first case of northern pike outside of its native range in BC. Since this time, several more mature pike have been captured from this lake but not in any of the inlet or outlet systems (J. Burrows, pers. comm.).

Distribution Trends.— Until very recently, the spread of this species has been negligible, as indicated by only two recorded observations outside the native range of northern pike. However, the new finding in the Kootenay Region suggests that this trend may change.

Stocking Records for BC.— There are no records of authorized stocking of northern pike in British Columbia.

Mode of Invasion.— The northern pike has been introduced to many locations in Canada and the US in an effort to enhance recreational fishing opportunities. Currently most new introductions are unauthorized, also in an effort to develop fishing opportunities. In the past, pike have been released to control stunted populations of prey fish or to reduce densities of some species (e.g., carp and shad) in reservoirs (Fuller 2005c). Northern pike spread via natural dispersal, once introduced to new habitats. For example, they have spread throughout the lakes and rivers of the Suisitna watershed in Alaska following illegal introduction into a single lake (Fay 2002). This fish has been reported in the Flathead River in Montana, as well as Coeur d'Alene Lake in Idaho (Fuller et al. 1999), so there is the possibility of this species moving northward into BC waters. Although the Haha watershed drains into the trans-boundary Kookanusa Reservoir, the outlet is believed to flow below the surface, preventing upstream movement of reservoir fish. Thus, it is likely that pike in Haha Lake are the result of illegal introductions.

6.14.7 Threats

Pike are widely-known as voracious predators, particularly of other fish species, though there have been very few direct assessments of northern pike introductions. As top-level predators they are known to cause declines in prey species, sometimes to the point of local extinction (Findlay et al. 2000; Fay 2002). In an experimental introduction of northern pike, He and Kitchell (1990) found that small-bodied species were less susceptible to predation than larger-bodied species, and negative effects on these species were less pronounced. However, the authors were assessing the effects of only a single size class (relatively large pike), so this is likely not a generalization. Small pike would likely key on small-bodied prey, so effects may occur across all size classes of prey fish; indeed they have in some other studies (e.g., Findlay et al. 2000). In addition to direct effects on abundance, northern pike introduction can produce behavioural shifts in prey species causing changes in distribution (He and Kitchell 1990).

Most ecological changes attributed to northern pike are based on anecdotal information only. For example, stickleback populations are absent or rare in some Alaska lakes with introduced pike (Fuller 2005c), landlocked salmon are rarer and considered threatened following introduction of pike in some lakes in Maine (Brautigam 2001), and salmonid populations are thought to be threatened by northern pike introductions in the Pacific Northwest (McMahon and Bennett 1996). Northern pike are implicated in the decline of native westslope cutthroat trout (*Oncorhynchus clarki lewisi*) and bull trout (*Salvelinus confluentus*) in some Montana waters (McMahon and Bennett 1996). Pike are considered the fish species of greatest concern to parts of Alaska (Fay 2002), where they have been said to have caused "widespread measurable damage to resident species (primarily rainbow trout and grayling) and potential impact to coho salmon stocks in the Suisitna River drainage. Wherever pike are introduced, they eliminate or greatly reduce the native species present. They have the potential to cause severe environmental and economic impacts to many highly productive salmon streams." Although these statements are not supported with data, northern pike introductions are clearly a major concern to biologists in the Alaska Department of Fish and Game. In Montana, northern pike commonly deplete prey sources and then become stunted (McMahon and Bennett 1996 in Fuller et al. 1999).

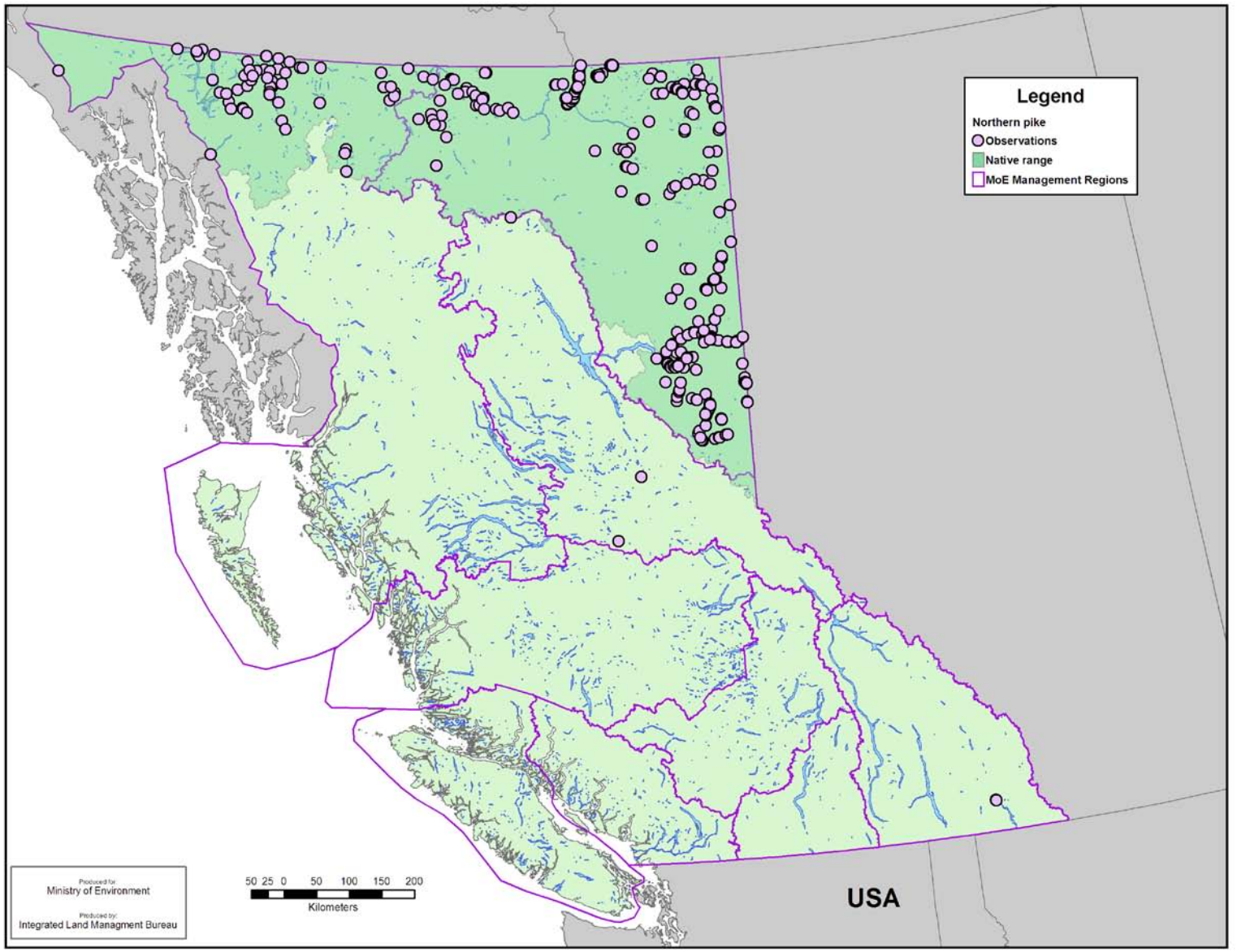


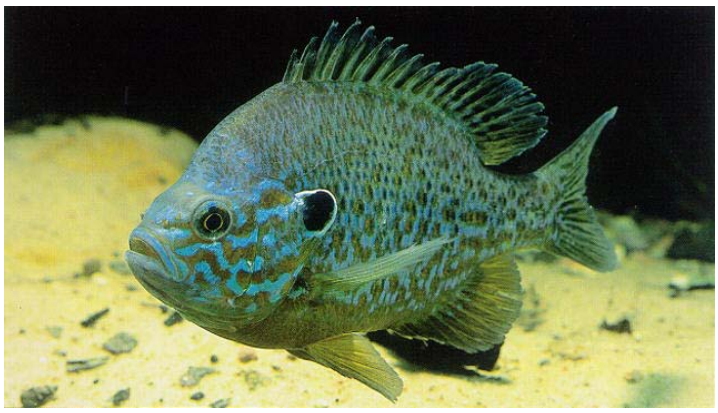
Figure 25. Northern pike occurrences in British Columbia, based on the MoE occurrences database, and the assumed native range of the species. Most records occur within the species' native range.

6.14.8 Potential Management Actions

Northern pike are difficult to eradicate once established. Efforts are likely best-spent on trying to prevent human-aided dispersal of this species into new habitats. Information on northern pike should be included in material used to educate the public on the ecological dangers of translocating wildlife. Since virtually all pike translocations are motivated by desire for enhanced fishing opportunities, education efforts should be targeted primarily at anglers.

It is possible to control or even eradicate northern pike from some systems, but efficacy of efforts will be dictated by the natural features of the waterbody (e.g., still vs. running water, size, depth, pH, etc.), as well as social and economic factors. Control methods are discussed in detail in Section 5. In some systems, such as large lakes with existing populations of other desirable fish species, or resident human population it may be impossible to extirpate pike once they are established. In smaller systems it may be possible to use piscicides, but there are a number of ethical and social issues that preclude the widespread use of such methods. Where complete eradication is not possible, it may be possible to reduce pike numbers using physical removal with minnow traps, nets, and electrofishing, but such efforts would need to be continuous to be effective at controlling numbers, and attempts have failed in the past to produce desired effects (California Department of Fish and Game 2004).

6.15 Pumpkinseed (*Lepomis gibbosus*)[6]



6.15.1 Taxonomy

Family: Centrarchidae (Sunfishes)
Genus: Lepomis
Species: gibbosus
Authority: Linnaeus, 1758

6.15.2 Habitat

Pumpkinseed favour littoral areas for rearing, and are usually found in small lakes, ponds, or shallow, weedy bays of larger lakes (Scott and Crossman 1973). Although primarily a lake-dwelling species, they may also be found in the quiet areas of slow-moving streams (Scott and Crossman 1973). They prefer clear water and the cover of submerged or aquatic vegetation, where they often aggregate in large numbers (Scott and Crossman 1973). In warm waterbodies of southern Ontario, this species is often numerically dominant (Scott and Crossman 1973).

Spawning Habitat.— Nests are built in shallow littoral areas, usually in areas of submerged aquatic vegetation (Scott and Crossman 1973). Bottom substrates are variable, including clay, sand, gravel or rock (Scott and Crossman 1973).

6.15.3 Diet

Pumpkinseed are primarily insectivores, though many other invertebrates are common in their diets. Larger pumpkinseed may feed on small fishes and other vertebrates, such as salamanders (Scott and Crossman 1973). Food is taken from the benthos, water column, and water surface (Scott and Crossman 1973).

6.15.4 Life History

Reproduction.— Spawning begins in late spring and early summer and may continue into mid- or even late-summer (Scott and Crossman 1973). Males construct nests in shallow water,

[6] Image used with permission. Copyright Steffen Zienert.

usually in 15 to 30 cm depth, beginning when water temperatures reach about 20 ˚C (Scott and Crossman 1973). There is a considerable courtship display and ritual. Males may spawn multiple times in one season, and with more than one female (Scott and Crossman 1973).

Age at First Reproduction.— Sexual maturity is usually attained by 2 years (Scott and Crossman 1973).

Life Span.— Scott and Crossman (1973) list life span in Canada as 8 to 10 years.

Body Size.— Mature pumpkinseed are usually 18 to 23 cm in length; maximum size in Ontario is thought to be 25 cm, 480 g (Scott and Crossman 1973). FishBase (2005) lists maximum size as 40 cm and 630 g.

Fecundity.— Egg number increases with body size. For females 2 to 5 years and 61 to 92 mm, egg number was measured as 600 to 2923, with an average of 1,684 to 2,923 (Ulrey et al. 1938 cited in Scott and Crossman 1973). In other studies, egg number has been cited as being as high as 5,000 per female (Scott and Crossman 1973).

6.15.5 Human Interest

Pumpkinseed sunfish are a common target for anglers, although they are not normally trophy fish and are sometimes considered an undesirable catch item (FishBase 2005). They are readily caught on a variety of lures and bait (Wooding 1994). In some areas of Ontario and Quebec, pumpkinseed are part of the commercial fishery take, and they are considered good to eat (Scott and Crossman 1973). Because young pumpkinseed are common in the diets of many predatory fishes (Scott and Crossman 1973) they are sometimes translocated to form part of a forage base. Young pumpkinseed are occasional aquarium fish, but are often only short-term residents due to their fast growth rate (Scott and Crossman 1973).

6.15.6 Distribution

Native Distribution.— The natural distribution of pumpkinseed is the freshwaters of eastern North America. This distribution extends from the Great Lakes and southern St. Lawrence area (excluding Lake Superior) and southeastern Manitoba, south to Illinois, Kentucky and Georgia (Scott and Crossman 1973; NatureServe 2005). In Canada, the species is found in southeastern Manitoba and western Ontario in the Winnipeg River system. Further east it is found in the Great Lakes and St. Lawrence region of southern Ontario and Quebec, with some populations also found in southern New Brunswick.

Present Distribution.— Pumpkinseed sunfish have been successfully translocated to many regions of North America, Europe, Russia, Africa and South America (FishBase 2005). In North America, pumpkinseed are now naturalized in most western US states, and southern British Columbia (Scott and Crossman 1973; Fuller 2005d; NatureServe 2005). In adjacent Washington, pumpkinseed are widely distributed in the Puget Sound area and the Columbia system, in a variety of lakes, ponds and reservoirs (Wydoski and Whitney 2003).

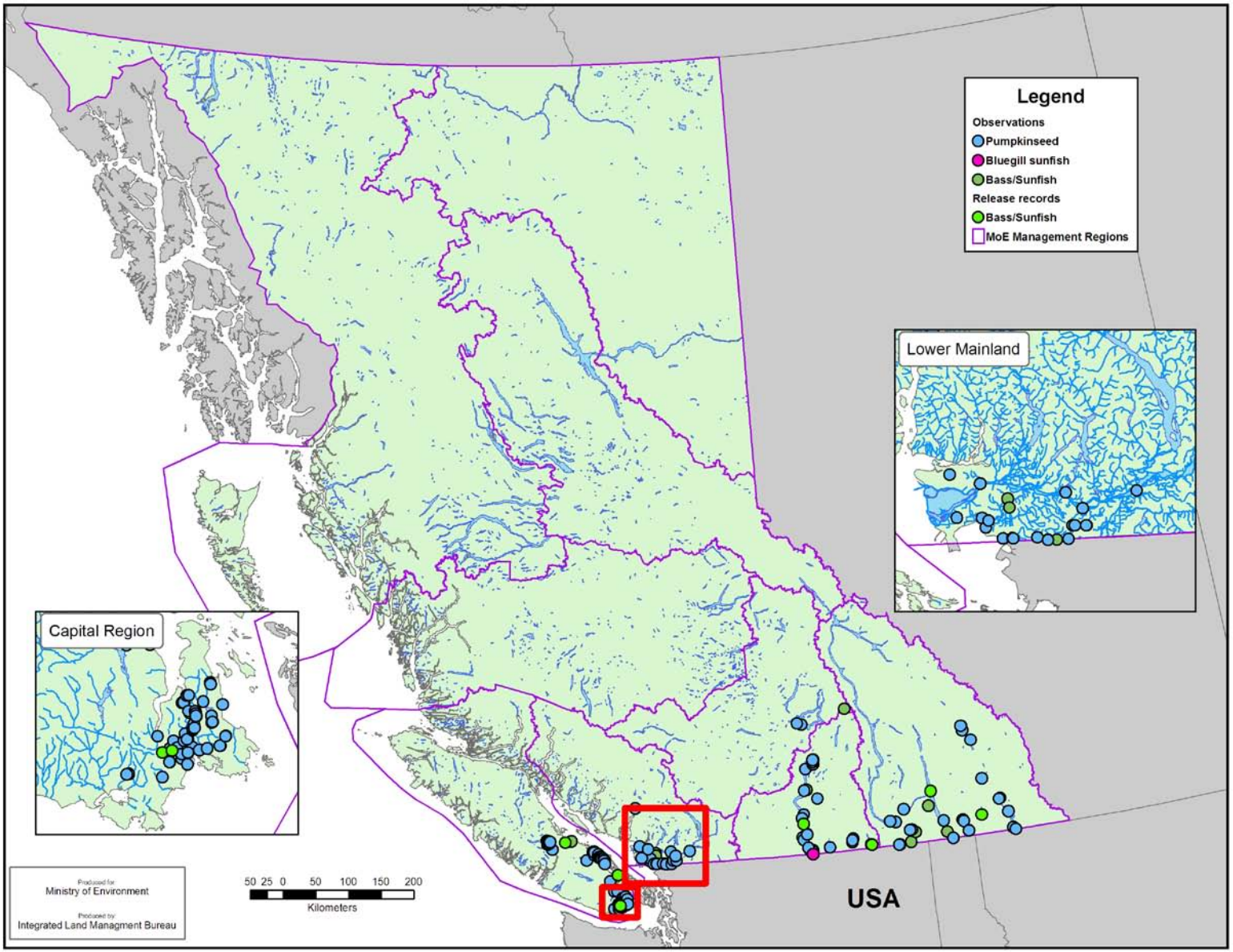


Figure 26. Sunfish occurrences in British Columbia, based on the MoE occurrences database. Note that some specimens were not identified to species, and are tracked simply as “bass / sunfish.” Also shown on this map are the sites for authorized releases.

BC Distribution.— There are records for pumpkinseed in a total of 86 waterbodies in BC, spread over four of the nine MoE regions. The following table summarizes unique waterbody occurrences within MoE regional boundaries, based on records in the occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Region	No. of waterbodies
1	43
2	21
3	-
4	12
5	-
6	-
7	-
8	11
9	-

Observations to date in BC are limited to southern regions, likely due to warm water requirements, as well as proximity to urban areas and source populations in the US.

Distribution Trends.— Although many of the observations are undated, the data indicate a considerable rise in the number of waterbodies with pumpkinseed. Most observations have occurred in the last 30 years.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
1	19	66	86

Stocking Records for BC.— There are no specific records of authorized stocking of pumpkinseed in British Columbia. However, there are nine records marked as "bass/sunfish (general)," which date to the early 1900s. These introductions occurred in seven lakes in the Vancouver Island, Kootenay and Okanagan Regions, listed in the following table.

Waterbody	**MoE Region**	**Year**
St. Mary Lake	1	1920
Langford Lake	1	1901
Florence Lake	1	1901
Spider Lake	1	1923
Moyie Lake	4	1901
Fraser Lake	4	1923
Christina Lake	8	1901

It has been assumed that these introductions were of smallmouth bass, but it is possible that stocks were pumpkinseed, or mixed with pumpkinseed. Of these lakes, there are observations in the occurrences database of pumpkinseed in Langford Lake, Florence Lake, Christina Lake, as well as Moyie River. However, it is clear that the great majority of introductions have been unauthorized.

Mode of Invasion.— Historically, pumpkinseed were stocked in many non-native watersheds to enhance sportfishing opportunities elsewhere in North America (Fuller 2005d). In some instances, the species spread further due to natural dispersal. For example, Scott and Crossman (1973) attribute the presence of pumpkinseed in the Southern Interior of BC to dispersal from non-native populations in the Columbia River system. As occasional aquarium fish, they may at times be spread by release of pet stock into the wild. It is unknown how the sunfish arrived on Vancouver Island, and whether introductions are historic only or continuing to occur via illegal movement or natural spread. As noted above, historic stocking of "bass/sunfish" occurred in only a few locations, and it is unclear if any of these activities used sunfish.

6.15.7 Threats

The impacts of non-native pumpkinseed on native biota have not been widely studied or reported, so ecological effects are not well understood. In general, there is exceedingly little scientific or even anecdotal evidence of adverse effects from the introduction of this species. A search of multiple web-based databases on invasive species produced little more than mention of the species as an occasional (and unspecified) pest. This in itself may indicate that impacts to native biodiversity are usually not severe.

The pumpkinseed is known to hybridize with a number of other sunfish species (Scott and Crossman 1973), and this may have impacts on native sunfishes. This is not a concern in BC, since there are no native sunfishes.

In an effort to assess the impact of non-native pumpkinseed, Lodge et al. (1998) studied their effect on the abundance and diversity of aquatic snails, a common diet item. Laboratory and field experiments suggested there would be an effect, but this could not be demonstrated in a survey of lakes in northern Wisconsin. Bonar et al. (2005) examined stomach contents of several non-native species, including pumpkinseed, in shallow lakes in Washington, to see the effect of non-native species on coho smolt production. Predation by pumpkinseed on coho smolts was negligible.

The closely related green sunfish (*Lepomis cyanellus*) was recently observed in Osoyoos Lake (D. McPhail, pers. comm.). Green sunfish appear to be at least partially responsible for the decline or loss of a number of native fishes and some amphibians in the US due to aggressive competition and predation (Fuller et al. 1999 and references cited therein). In cold lakes, they often form dense populations and reduce trout populations (McKechnie and Tharratt 1966 cited in Fuller et al. 1999).

6.15.8 Potential Management Actions

There is little to distinguish the management options of pumpkinseed from other non-native fish species: it may be possible to catch them in large numbers using minnow traps, nets, and electrofishing, but extirpating a population of sunfish is likely to require the coincident use of several of the methods discussed in Section 5. In some systems, such as large lakes with existing populations of other desirable fish species it would likely be impossible to extirpate pumpkinseed once they are established. In smaller systems it may be possible to use piscicides,

but there are a number of ethical and social issues that preclude the widespread use of such methods.

Efforts are therefore best spent on preventing introductions to habitats that do not currently support sunfish. These efforts will require regulation, enforcement, education and stewardship. In theory, adequate statutes and regulations already exist: importation of any sunfish species is prohibited, as is the possession, transport and introduction of any live fish into natural waters without a permit. (see Section 2.2) This should limit attempts to release this and other fish species, although it may be difficult to enforce this effectively. Education is most needed for those that are likely to translocate these (or other) fish. Strategies should be developed to ascertain who is likely to introduce fish, and to target communication at such groups and individuals. Strategies should also be developed that would aid public stewardship of freshwater bodies in BC, to prevent the introduction and spread of aquatic species in BC.

6.16 Smallmouth Bass (*Micropterus dolomieui*)



6.16.1 Taxonomy

Family: Centrarchidae (Sunfishes)
Genus: Micropterus
Species: dolomieui
Authority: Lacepède, 1802

6.16.2 Habitat

Habitat is variable and varies with body size and season. During the spring, fish congregate on the spawning grounds but later disperses to rocky or sandy areas in lakes in fairly shallow water, and in greater depths during the height of summer (Scott and Crossman 1973). Smallmouth bass are generally found in proximity to rocks shoals and talus slopes, or logs, and are less frequently associated with dense aquatic vegetation than largemouth bass (Scott and Crossman 1973). They prefer lower water temperatures than largemouth bass (Scott and Crossman 1973), but temperature preferences are still considered relatively warm compared to most native fish in BC. These fish are relatively inactive during winter months when they remain near the bottom; feeding in the spring begins when temperatures reach 8.5 °C (Keast 1968 cited in Scott and Crossman 1973).

Juvenile Habitat.— Cover is used for resting, escape, and feeding activities primarily to reduce predation risk from piscivorous fish, birds, amphibians, and other predators (Scott and Crossman 1973; Olson et al. 2003). Cover provides secure areas within which fish can rest with minimal risk of predation, and is used by juveniles as a place of refuge during feeding. Cover can take the form of submerged woody debris, interstitial spaces within areas of cobble or boulder substrate, or the stems of aquatic vegetation (Scott and Crossman 1973). Distance between feeding fish and cover is an indication of how secure the fish are from predation during any particular time, as well as the relative abundance of food at different distances from cover. During the first few weeks after their emergence juveniles are extremely vulnerable to predation, and thus require forage at or close to their primary source of cover (Sabo and Orth 1994). Areas of aquatic macrophyte growth are heavily exploited by juvenile smallmouth bass during this period. At older stages juvenile bass will rely less upon macrophytes, and instead

aggregate in schools during feeding activity to reduce predation risk (Sabo and Orth 1994). Substrate composition is reported to be predictive of juvenile habitat quality, with coarse substrates of gravel, cobble and boulders preferred over areas where silt or organics are the dominant substrate type.

Adult Habitat.— Adult smallmouth bass are predominantly piscivorous and cruise open water and littoral areas in search of prey. As "warmwater" fish, smallmouth bass are reported to prefer a temperature range of 20 to 26 °C during the summer months, and largely remain above the thermocline. The upper lethal limit for these fish is reported as 35 °C (Scott and Crossman 1973).

Spawning Habitat.— Males provide all parental care (Scott and Crossman 1973; Hinch and Collins 1991). In the spring, they acquire territories in the littoral region where they build nests and mate (sometimes with many females, Weigmann 1992). Nests are built in areas sheltered from wind and wave action (Scott and Crossman 1973; Lukas and Orth 1995). Preferred water depths range from 1m to 5m, with the upper portion of this range most preferred (Scott and Crossman 1973). Spawning substrate preference is reported as gravel/cobble mixture, followed by silt or mud, boulder or rock, and finally detritus and vegetation if other substrates are unavailable (Scott and Crossman 1973; McNeill 1995). Shoal areas with large boulders and gravel are also reported as good spawning habitat. Gently sloping bathymetry is preferred over steeper areas. There is a strong tendency for males to return to within 20 m of their previous year's nest site, and over 80% of male spawners renest within 200 m of their previous nest site (Scott and Crossman 1973; Ridgway et al. 1991).

6.16.3 Diet

Juvenile Diets.— Smallmouth bass utilize different types of forage as they increase in size during their first year (Scott and Crossman 1973). During the first two to five weeks after swim up (depending on their growth rate) young fish are planktivorous, feeding on zooplankton (Scott and Crossman 1973). The diet of juveniles shifts from plankton to immature aquatic, terrestrial drift, and benthic invertebrates as the fish reach about 20mm in total length, then fish and crustacea replace invertebrates as the dominant prey type when juveniles reach approximately 50 mm in total length (Scott and Crossman 1973; Olson et al. 2003). Tadpoles, fish and amphibian eggs, and plant material are reported as forage items as well (Scott and Crossman 1973).

Adult Diets.— Adult smallmouth bass are predominantly piscivorous, but will also include insects and crayfish, as well as frogs and fish eggs, in their diets (Scott and Crossman 1973).

6.16.4 Life History

Reproduction.— Smallmouth bass spawning occurs in the spring, and is initiated by an increase in water temperature. Males provide all parental care. In the spring, they acquire territories in the littoral region where they build nests and mate (sometimes with many females). Nest construction may begin at 13°C, and egg deposition is reported to occur when water temperatures are between 16°C and 18°C. Male smallmouth bass defend juveniles within the nest area for approximately two to three weeks post-hatching, but gradually decrease their

protective behaviour, finally abandoning the nest at the end of this period. At this point the young leave the nest area and seek out appropriate rearing habitat.

Age at First Reproduction.— Sexual maturity for males occurs at 3 to 5 years, and 4 to 6 years for females (Scott and Crossman 1973).

Life Span.— Maximum age in Canada is about 15 years (Scott and Crossman 1973). Maximum reported age is 26 years (FishBase 2005).

Body Size.— Maximum recorded size is 69 cm TL and 5.41 kg (FishBase 2005). Typical sizes are considerably smaller, and Scott and Crossman (1973) note that fish of 2.2 to 3.2 kg are usual prize-winning sizes in angler contests in Canada.

Fecundity.— Egg number in females is size-dependent, and ranges from 5,000 to 14,000, or approximately 7,000 eggs per pound (454 g) of female (Scott and Crossman 1973).

6.16.5 Human Interest

Smallmouth bass are held in high esteem by many anglers for their sporting quality and tasty flesh. Attractiveness to anglers is often cited as the primary reason for the spread of this species throughout its non-native range. Smallmouth bass were widely stocked throughout North America by government fish and wildlife agencies, but the species is most commonly stocked now (illegally) by anglers wishing to enhance sport fishing opportunities (Rahel 2000).

6.16.6 Distribution

Native Distribution.— Smallmouth bass are native east of the North American continental divide; the distribution is described in Scott and Crossman (1973) as "fresh waters of eastern central North America." A FishBase query described the native distribution as: St. Lawrence-Great Lakes system, Hudson Bay and Mississippi River basins from southern Quebec in Canada to North Dakota and south to northern Alabama and eastern Oklahoma in the USA (FishBase 2005).

Present Distribution.— Smallmouth bass are widely distributed, with self-sustaining populations listed in the following countries and regions: UK, Europe, Russia, Middle East, Northern and Southern Africa, Continental US, Southern Canada, Caribbean, South America, Asia, Southeast Asia, Hawaii, Mauritius, Madagascar, Fiji, Guam, and New Caledonia (FishBase 2005; ISSG 2005). Smallmouth bass are common in lowland lakes and ponds in the Puget Sound area of Washington, and throughout the Columbia system, most notably in reservoirs (Wydoski and Whitney 2003).

BC Distribution.— The following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. Note that since some waterbodies span more than one MoE region, the total number of unique waterbodies in the province may be different than the sum of counts within each region. Most recently (July 2006), five individuals were captured in the Beaver Creek system, one of which was only 200 m upstream of the Quesnel River. These findings represent the first

documentation in the Cariboo Region and a significant threat to anadromous waters. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Region	No. of waterbodies with "smallmouth bass"	No. of waterbodies with "bass/sunfish - general"	No. of waterbodies with one or both
1	37	5	37
2	1	5	6
3	2	1	3
4	4	7	9
5	1	-	-
6	-	-	-
7	-	-	-
8	7	3	10
9	-	-	-

The database tracks occurrences of smallmouth and largemouth bass separately, but there is also a third category of "bass/sunfish - general," for observations that do not or cannot distinguish between the two bass species and/or sunfish species. We assume that at least some of these observations are of smallmouth bass.

Smallmouth bass are not native to British Columbia, but the data indicate they are now distributed widely in southern portions of the province, particularly on Vancouver Island and the Okanagan. Observations in the Okanagan are likely undercounted due to redundancy in the watershed codes on which database queries were made (see database description in Section 6.1).

Distribution Trends.— The following data come from the occurrences database, and indicate an increasing trend in the number of waterbodies with smallmouth bass. The trend toward increasing numbers of waterbodies with smallmouth bass is likely accurate, but the data themselves should not be used to estimate a rate of change in the species distribution. Biological surveys are often conducted opportunistically in different places for a variety of purposes. Smallmouth bass were undoubtedly established in lakes prior to 1950, but surveys had either not been conducted or the records were not incorporated into the database. Note that some records in the database do not have years associated with the record, so the total number of waterbodies exceeds the value in 2000, when the last survey record was posted for this species.

	Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
smallmouth bass	2	19	33	50
Bass /sunfish	-	1	6	21

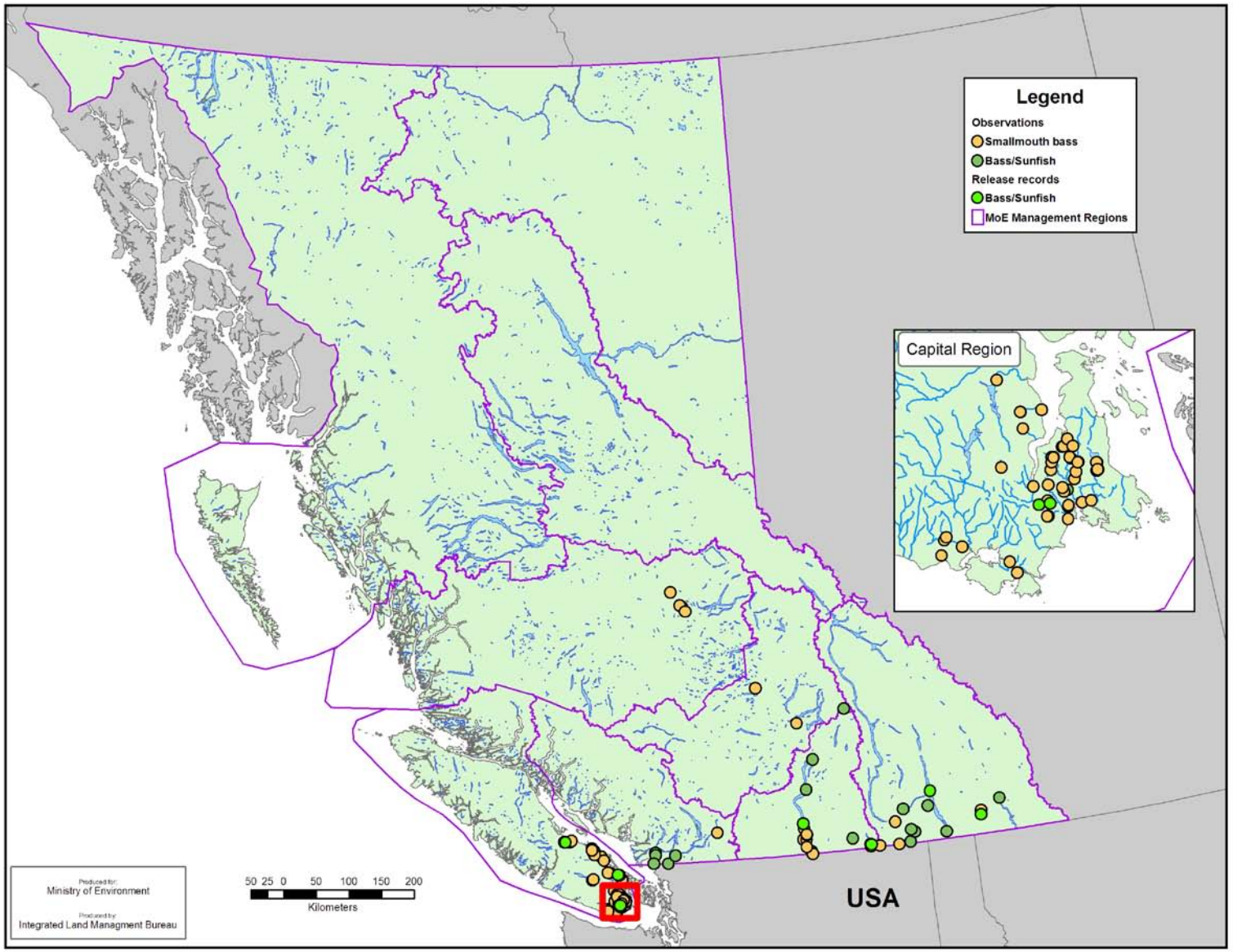


Figure 27. Smallmouth bass occurrences in British Columbia, based on the MoE occurrences database. Note that some specimens were not identified to species, and are tracked simply as "bass / sunfish." Also shown on this map are the sites for authorized releases.

Stocking Records for BC.— There are no specific records in the releases database of authorized stocking of largemouth bass in British Columbia. However, there are seven records marked as "bass/sunfish (general)," which date to the early 1900s. These introductions occurred in four lakes on Vancouver Island, two in the Kootenay Region and one in the Okanagan Region, as listed in the following table.

Waterbody	MoE Region	Year
St. Mary Lake	1	1920
Langford Lake	1	1901
Florence Lake	1	1901
Spider Lake	1	1923
Moyie Lake	4	1901
Fraser Lake	4	1923
Rykerts Lake*	4	1980's
Tie Lake*	4	1980's
Christina Lake	8	1901
Skaha Lake*	8	1987
Vaseux Lake*	8	1987

*Confirmed smallmouth bass (see text below)

It has been assumed that these introductions were of smallmouth bass, but it is possible that stocks were largemouth bass, pumpkinseed sunfish, or a mixture. Of these lakes, only Fraser Lake does not have an independent species-specific observation of smallmouth bass, so most of these early introductions appear to have been successful. Finally, smallmouth bass were introduced by regional staff into Skaha and Vaseux lakes in Region 8 in 1987 (B. Jantz, pers.comm.), and from Duck Lake to Rykerts Lake and from Manistee Lake to Tie Lake in Region 4 in the mid-1980's (G. Oliver, pers. comm.), although these records are not included in the release records database. However, it is clear that the great majority of introductions have been unauthorized.

Mode of Invasion.— The initial release and spread of smallmouth bass in BC and in the US was facilitated by government-sponsored programs that were mandated with enhancement of angling opportunities. This species was stocked historically in only a few locations in BC, but some protection of smallmouth bass is actively promoted through angling regulations. The mode of spread at present seems to be individuals or groups seeking to enhance sportfishing opportunities in some regions of the province. The procedures used by these groups to transfer fish among lakes likely vary, but it is presumably not difficult. However, it is impossible to determine if the current distribution of smallmouth bass in BC is the result of relatively few 'successful' introductions and a large number of 'failed' attempts, or a high success rate in a limited number of systems. Unaided transfer to downstream lakes and reservoirs (and occasionally to large low gradient river) is possible when bass become established and disperse downstream, but the likelihood of downstream transport depends on a number of factors including distance to favourable habitats, physical characteristics of the receiving and migration habitats, and the number of emigrants from a system. Unaided upstream dispersal is possible, but is likely uncommon in BC due to the relatively high gradient of streams in the province. Smallmouth bass are classified as a "warmwater" species and as such are limited by cold

temperatures. Under global warming projections suitable habitat for warmwater fish species is expected to increase markedly (Stefan et al. 2001), with range expansions possible as a result.

6.16.7 Threats

The primary documented threats of smallmouth bass are direct effects of predation on small-bodied native fish species and indirect effects through alteration of aquatic food webs.

There is plentiful evidence of the effects of smallmouth bass predation on the abundance and distribution of littoral minnow species in small eastern lakes. Most of this evidence is correlational through studies of species associations, rather than experimental. Reduced abundance, altered habitat use, and local extirpation of small-bodied littoral minnow species is common in the presence of non-native piscivores, especially smallmouth bass. Chapleau et al. (1997) looked at fish communities of small lakes (< 61 ha) in Gatineau Park, Quebec, which have had piscivorous fish introductions during the past 85 years. They found a significant reduction of species richness of small-bodied fish species in lakes with non-native piscivores. Five native small-bodied species were found only in lakes lacking piscivores, and the authors predicted that local extinction would occur if piscivores are introduced to these lakes. Similar studies have been conducted in other areas of eastern Canada and the US with similar results (e.g., Whittier et al. 1997; Vander Zanden et al. 1999; Whittier and Kincaid 1999; Findlay et al. 2000; MacRae and Jackson 2001). McRae and Jackson (2001) found on average 2.3 fewer small-bodied species in lakes with smallmouth bass, than lakes without smallmouth bass. In some cases, native brook trout were also deemed to be at high risk from introductions of smallmouth bass (Whittier and Kincaid 1999).

Smallmouth bass have been implicated in population declines of some salmonid populations in western North America. For example, Fritts and Pearsons (2004) found that smallmouth bass ate up to 85% of naturally-spawned ocean type chinook in the Yakima River, Washington. On closer inspection bass have been confirmed as significant predators of salmonids, but they have not been found to be the primary cause of population declines (Zimmerman 1999; Fayram and Sibley 2000; Fritts and Pearsons 2004). In some cases, the impact of smallmouth bass predation on salmonid populations may be mitigated by their different temperature and habitat requirements, which can reduce encounter rates (Fayram and Sibley 2000).

Food web effects by smallmouth bass have been documented in several major studies. Vander Zanden et al. (1999) used stable isotope analysis to study food webs in small lakes in southern Ontario. The lakes naturally had a simple fish community dominated by littoral minnow species and lake trout (*Salvelinus namaycush*), the only piscivore. Smallmouth bass were not native in this region, but have become established in many lakes. Vander Zanden et al. (1999) found that in lakes without smallmouth bass, lake trout fed predominantly on littoral minnows; in lakes with smallmouth bass lake trout fed predominantly on zooplankton in the pelagic zone. In other words, non-native smallmouth bass essentially took over the resource that was previously the sole domain of lake trout. From this study it was not clear whether the mechanism is one of competitive exclusion of lake trout or agonistic interactions between bass and trout (or both), but sampling indicated a substantial decline in abundance of littoral prey fish in lakes with smallmouth bass. In an expanded study, Vander Zanden et al. (2004) found that lake trout are buffered from the effects of smallmouth bass in lakes that also have abundant

pelagic fish prey; since many lakes in their study region do have pelagic fish prey species, lake trout were projected to be "highly vulnerable" in only 6% of lakes in which they currently reside. Bass may, however, have other effects on the lake ecosystems.

In a different study of food web effects, Power (1990) and Power et al. (1985) examined the effect of bass presence on algae abundance in low gradient streams in Oklahoma. Using experimental enclosures they found that bass presence had a substantial effect on algae abundance as mediated through grazing by minnow species. In the presence of smallmouth bass, algae growth was significantly greater than in the absence of smallmouth bass. Smallmouth bass influenced the behaviour and abundance of the grazing minnow with concomitant effects on algae growth. These effects were detectable over the fairly short duration of the experiments of 10-13 days.

Effects of smallmouth bass introductions are apparently not restricted to fish species. Kiesecker and Blaustein (1998) found that smallmouth bass interacted with non-native bullfrog to have a significant effect on habitat use, growth, and survival of native red-legged frogs in an Oregon-based experimental study. Smallmouth bass on their own had little effect, and the mechanism for the interactive effects was not known. This species may be partially responsible for the decline of native ranid frogs and the tiger salamander in California (Fuller et al. 1999 and references cited therein).

In BC, regional biologists report that in small lakes where smallmouth bass have been introduced, the survival of stocked salmonid fry is negligible (T. Yesaki, pers. comm.), with the result being loss of highly-productive small lake trout fisheries. No equivalent data is available for impacts on wild populations of salmonids in these lakes so it is impossible at this time to determine if the impacts are due to the naïveté of hatchery fish to predators. However, given the piscivorous habits of smallmouth bass, its presence is likely to have an impact on survival of both wild and hatchery salmonid juveniles (see Bonar et al. 2005).

Threats Summary.— Smallmouth bass are a significant threat to aquatic biodiversity in BC, as well as to fisheries of various native species. Bass are attractive to anglers and have a history of being actively stocked (both legally and illegally) in BC and elsewhere in North America. Bass are top-level carnivores, the trophic level most likely to cause substantial ecological effects. The species has been shown to have substantial impacts on fish communities elsewhere in North America. Smallmouth bass are often classified as a "warmwater" species, but there is abundant habitat for the species in BC lakes, particularly in the southern portions of the province. The spread of this species would depend on active transport by humans; it is unlikely to disperse among watersheds on its own.

6.16.8 Potential Management Actions

It is noteworthy that during our review of smallmouth bass literature we came across almost no references of attempts at extirpation of non-native smallmouth bass populations. Almost all mention of management actions focussed on education as a means of slowing or preventing further spread of this species. Such efforts are probably best spent in areas that currently have both extant populations and suitable habitat that is not yet occupied. A single unpublished example of efforts to eradicate smallmouth bass via boat electroshocking comes from Little

Moose Lake in the Adirondacks. Intensive shocking efforts over three years reduced the bass population by over 90% and native fish abundance was seen to increase in the following year (Weidel et al. date unknown). However, long term success is unknown and it seems likely that the bass population would rebound once control pressures declined. A modelling exercise in Oregon demonstrated that even if bag limits are removed for bass species, harvest rates would only increase by 8% to 18% and have little effect on reducing predation effects on native salmonids (see Gorvindarajulu 2005).

Like largemouth bass, smallmouth bass are relatively easy to catch in large numbers using minnow traps, nets, and electrofishing, yet, extirpating a population of bass is likely to require the coincident use of several of the methods discussed in Section 5. In some systems, such as large lakes with existing populations of other desirable fish species it is essentially impossible to extirpate bass once they are established. Efforts are therefore best spent on preventing introductions to habitats that do not currently support bass. These efforts will require regulation, enforcement, education and stewardship. In theory, adequate statutes and regulations already exist: importation of bass is prohibited, as is the possession, transport and introduction of any live fish into natural waters without a permit (see Section 2.2). Unfortunately, none of these regulations are stringently enforced, and compliance is insufficient to prevent introductions. Education is most needed for persons who are likely to translocate these (or other) fish. Strategies should be developed to ascertain who is likely to introduce fish, and to target communication at such groups and individuals. Strategies should also be developed that would aid public stewardship of freshwater bodies in BC, to prevent the introduction and spread of aquatic species in BC. If Washington state is used as a reference point, there is clearly considerable potential (and presumably desire from some groups) to have smallmouth and largemouth bass spread in BC, and this represents a substantial threat to native fish biodiversity.

6.17 Tench (*Tinca tinca*)[7]



6.17.1 Taxonomy

Family: Cyprinidae (Minnows or carps)
Genus: Tinca
Species: tinca
Authority: Linnaeus, 1758

Note: Like the common carp and goldfish, tench have been domesticated for pond culture and different strains exist. However, most non-native populations of tench are of the natural European strain.

6.17.2 Habitat

Tench are benthic fish that prefer calm waters of ponds and low-gradient streams with mud substrates and abundant rooted aquatic vegetation (Scott and Crossman 1973; Gonzalez et al. 2000). Perrow et al. (1996) observed strong daytime associations with the littoral emergent vascular plant, *Typha angustifolia*, which grows in relatively deep water and forms stands of relatively widely spaced stems, allowing large fish to penetrate deeply into cover. Spawning occurs in shallow areas with abundant macrophytes (Scott and Crossman 1973). We found no mention in the literature of differential habitat use by juveniles and adults.

6.17.3 Diet

Juvenile Diets.— Tench feed on zooplankton while very young, and switch to a broad range of benthic invertebrates as they grow (Giles et al. 1990 cited in Brönmark 1994). Young tench also consume algae (Scott and Crossman 1973).

Adult Diets.— Tench feed primarily on a range of benthic invertebrates, especially insect larvae and molluscs (Scott and Crossman 1973; Giles et al. 1990 cited in Brönmark 1994; Brönmark 1994; Perrow et al. 1996). Tench have the capacity to crush mollusc shells; although molluscs

[7] Image used with permission. Copyright Steffen Zienert.

may not form a large portion of the diet in some cases, Brönmark (1994) found snails to be an important component of the diet of tench in experimental cages in a Swedish pond. He also found zooplankton to be an important component of their diet. Adamek et al. (2003) found zooplankton to be an important diet component when tench were in the presence of carp.

6.17.4 Life History

Reproduction.— In Europe, spawning occurs in June or July (Scott and Crossman 1973). Eggs are scattered on plants and rock in dense vegetation in shallow water (FishBase 2005). There is no parental care (FishBase 2005).

Age at First Reproduction.— Scott and Crossman (1973) note that tench are sexually mature within about four years.

Life Span.— Life expectancy of tench is 20-30 years (Scott and Crossman 1973).

Body Size.— Maximum size is listed as 84 cm and 7.5 kg (FishBase 2005), but tench are usually much smaller and any individual over 3kg is considered large in the UK (UK Angling 2005).

Fecundity.— Fecundity is high: 275,000 eggs per pound (450g) of female is cited as a typical fecundity (Scott and Crossman 1973).

6.17.5 Human Interest

Tench have been domesticated as ornamental and food fish for use in pond culture, but not to the same extent as carp or goldfish. The species is often used in conjunction with carp in aquaculture settings (Scott and Crossman 1973). Tench are largely ignored by commercial and recreational fishers in North America (Scott and Crossman 1973; Nico and Fuller 2005b), however, they have a devoted following by European anglers as evidenced by the many sites on the internet presenting advice and information on fishing for tench. Tench can be caught by baited still-fishing methods. There was widespread interest in stocking this species in the US during the late 1800s, but there is little interest in the species today (Nico and Fuller 2005b).

6.17.6 Distribution

Native Distribution.— Tench are native to flatland rivers and lakes of Europe, (including Britain and Ireland) and Asia (Scott and Crossman 1973). In Europe, tench occur to 61˚ N, and in Asia they are found in some Arctic Ocean drainages (Scott and Crossman 1973). Although usually found in freshwater, tench may also occur in brackish water (FishBase 2005).

Present Distribution.— In addition to its extensive native range, tench have been introduced to Australia, New Zealand, North America, southern Africa, South America, and elsewhere (FishBase 2005). In North America tench were widely introduced in the US and have been recorded in 38 states (Nico and Fuller 2005b). Some initially successful populations in the US appear to have declined or been extirpated, perhaps through interactions with native or other non-native cyprinid species (Nico and Fuller 2005b). In Idaho, tench are found in the Coeur d'Alene and Pend Oreille systems (Wydoski and Whitney 2003). In Washington, tench are

found in various parts of the Columbia system, especially in northeastern and northcentral portions of the state (Wydoski and Whitney 2003).

Tench are not widely distributed in Canada, but are well-established in the mainstem and several sub-basins of the Columbia River (Scott and Crossman 1973). They have been recently found in the Richelieu River, which drains Lake Champlain, Quebec, and there are fears that tench will spread from this source (Anonymous 2000).

BC Distribution.— The following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Region	No. of waterbodies
1	-
2	-
3	-
4	2
5	-
6	1
7	-
8	2
9	-

Tench are established in the Columbia River mainstem, and several sub-basins including Christina Lake, and the Okanagan River system (Scott and Crossman 1973). Observation records indicate that tench are not widespread either locally or provincially.

Distribution Trends.— There are only five records for this species. Although there is an increasing trend in the number of waterbodies in BC with tench, there are still very few waterbodies in total.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
-	1	3	5

Stocking Records for BC.— There are no records of authorized releases of tench in BC.

Mode of Invasion.— There was widespread interest in stocking this species for food and sport fishing in the US during the late 1800s, and the US Fish Commission provided 138,000 fish to 36 states (Nico and Fuller 2005b). Efforts to introduce the species there were largely finished by 1900 (Nico and Fuller 2005b). There appears to be little interest in the species today (in either the US or Canada) as a food or sportfish, and it is largely ignored. Although typically found in calm waters, distribution of tench has likely been enhanced through both downstream and upstream dispersal. Scott and Crossman (1973) suggest that tench populations in the Okanagan

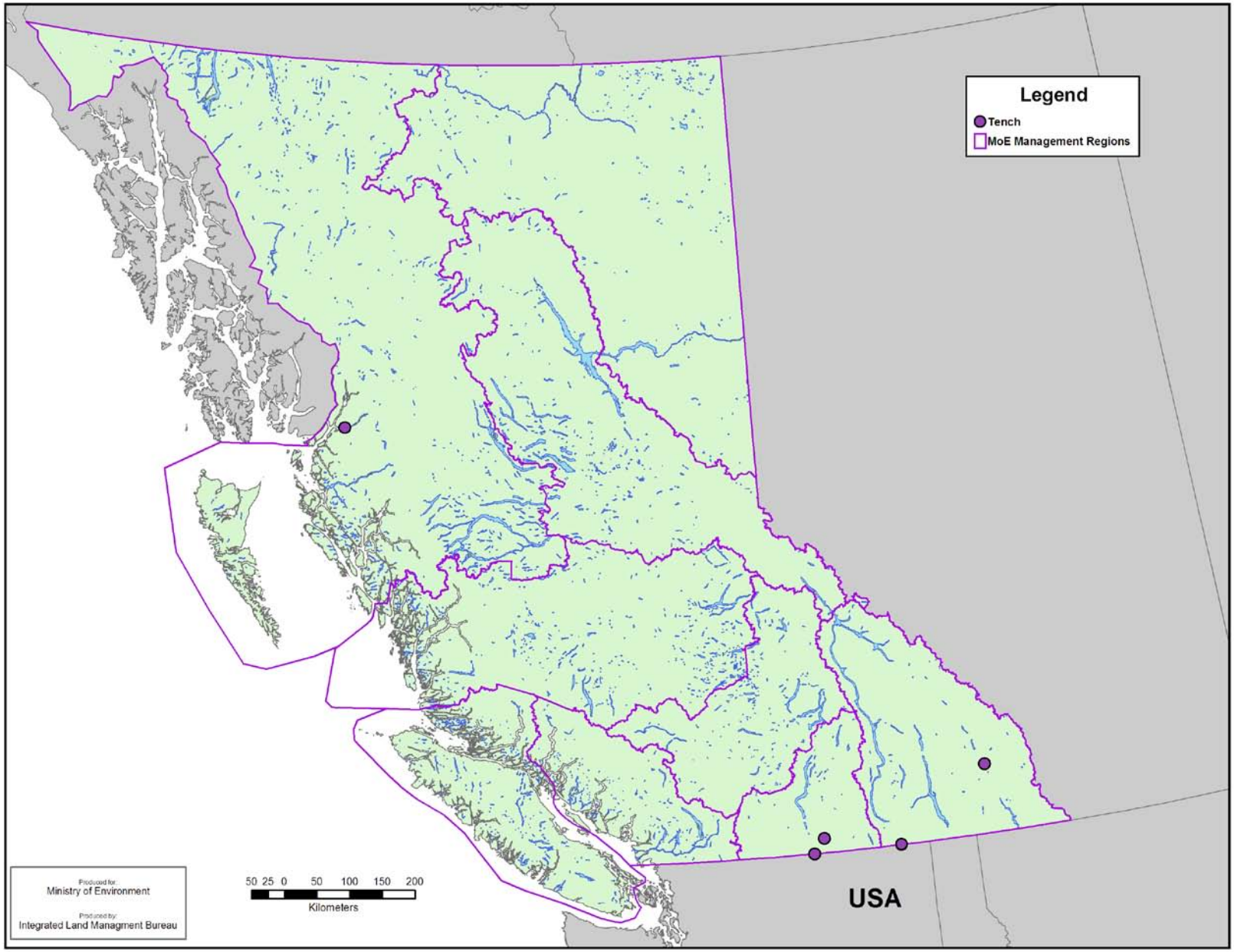


Figure 28. Tench occurrences in British Columbia, based on the MoE occurrences database.

River system likely derived from natural dispersal from populations in Washington State. The extent to which the species is still being introduced in BC is not known, but is presumably not great due to the species' limited appeal as a sportfish.

6.17.7 Threats

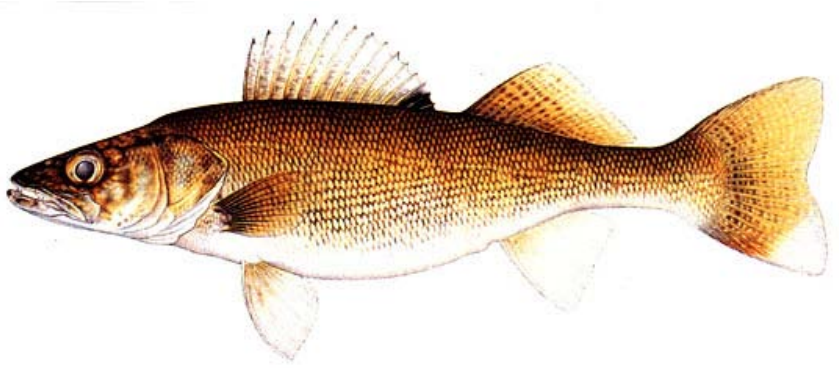
Very little is written about the ecological effects of tench as a non-native species. This may reflect a generally minor set of impacts to aquatic species and ecosystems. Kolar and Lodge (2002) conducted a risk assessment of non-native fish in North America and concluded that tench were not likely to develop into pest populations and likely to spread only slowly. Wydoski and Whitney (2003) found no evidence of ecological problems caused by tench. The most commonly cited effect of tench is an increase in turbidity, but even this impact is considered to be of smaller magnitude than that from other species such as carp. Indeed, Williams et al. (2002) found tench to have considerably less effect on macrophyte abundance in comparison to common carp. The mechanisms by which benthivorous fish increase turbidity are discussed in greater detail in Section 6.10.7. High turbidity can negatively affect visual predators, and can negatively alter water quality for drinking, agriculture and aesthetics. Moyle (1976 cited in Fuller et al. 1999) considered tench to be a potential competitor for food with native cyprinids and sport fish species.

The native distribution of tench includes northerly continental rivers and lakes, so there appears to be some potential for this species to spread into other areas in BC. However, the potentially high availability of habitat is likely offset by the species' limited appeal as a sportfish. If the status of tench were to increase there would likely be a greater threat from unauthorized introductions.

6.17.8 Potential Management Actions

Like most fish species, tench are likely difficult to eradicate once established, and efforts are probably best-spent on trying to prevent human-aided dispersal of this species into new habitats. Information on this species should be included in material used to educate the public on the ecological dangers of translocating wildlife. To our knowledge this species is not a particularly popular pondfish, but there should be some efforts to educate the public on the ecological consequences of releasing domesticated varieties of fish such as tench and other species.

6.18 Walleye (*Sander vitreus*)[8]



6.18.1 Taxonomy

Family: Percidae (Perches)
Genus: Sander
Species: vitreus
Authority: Mitchill 1818

Note: This species was formerly known as *Stizostedion vitreum*.

6.18.2 Habitat

Although they often spawn in tributary streams, walleye are predominantly lake-dwelling and do best in large, shallow, turbid lakes. Deep, turbid streams are also suitable (Scott and Crossman 1973). Eggs generally hatch in 12 to 18 days, and the yolk sac is absorbed rapidly after this. Within 5 days of hatching young begin to feed, and within 10 to 15 days of hatching young disperse into the upper levels of open water (Scott and Crossman 1973). By the latter part of the summer, young of the year move toward the bottom and are found in 3 to 9 m of water (Scott and Crossman 1973). Habitat suitability for adults is usually defined based on light intensity and temperature (Scott and Crossman 1973; Chu et al. 2004). Although they may do well in clear water, they tend to avoid areas of bright daylight, and feed only in the early morning and dusk (Scott and Crossman 1973). In turbid water walleye may feed continuously in the day (Scott and Crossman 1973; Chu et al. 2004). Optimal foraging temperatures are approximately 20 to 21 ˚C (Chu et al. 2004). Nate et al. (2003) classified lakes in northern Wisconsin based on physical characteristics, and found that lakes with walleyes were larger, deeper, and had more sand and gravel bottom than lakes without walleyes, which were smaller, shallower, and had more muck bottom; however, the range of lake characteristics was not large.

Spawning Habitat.— Walleye populations vary in the type of spawning habitat used as well as migration distances to reach historic spawning areas. Some populations utilize rocky, white water areas in rivers, whereas others spawn exclusively on rocky reefs in lakes (Jennings et al. 1996). There is evidence that walleye home to natal spawning areas (Scott and Crossman 1973; Jennings et al. 1996).

[8] Image used with permission from Robert Rice, www.nativefish.org

6.18.3 Diet

Diets during early life are dominated by invertebrates and fry of fish, including their own species (Scott and Crossman 1973); some populations are highly cannibalistic through most size classes (Chevalier 1973). The preferred food of post-larvae is crustacean zooplankton, with size selection of prey being more important than species (Mathias and Li 1982). During the juvenile stage, food preference shift from zooplankton to larger prey; first to *Chaoborus* and mayflies, then to *Gammarus* and fish (Mathias and Li 1982). Predation on smaller fish (including cannibalism) was observed at a total length of about 30 mm (Mathias and Li 1982). Juveniles and adults are primarily piscivorous, though insects may at times also be an important food (Scott and Crossman 1973; Lyons and Magnuson 1987; Baccante and Reid 1988). Predation by walleye can be a strong influence on fish populations (Forney 1974; Lyons and Magnuson 1987; Galinat et al. 2002; Schneider and Lockwood 2002; Baldwin et al. 2003).

6.18.4 Life History

Reproduction.— Walleye spawn in spring or early summer, depending on latitude and water temperature (Scott and Crossman 1973). Spawning may occur in water temperatures of 5.6 to 11.1 ˚C, but generally occurs during a narrower range of 6.7 to 8.9 ˚C (Scott and Crossman 1973). Spawning usually takes place in rapids below impassable barriers in rivers, or in shoal areas of lakes (Scott and Crossman 1973). Males arrive at the spawning grounds first; they are not territorial and do not build nests (Scott and Crossman 1973). Spawning occurs at night, in groups with one or two females and two to six males (Scott and Crossman 1973). Eggs are broadcast and settle into interstitial spaces in the substrate (Scott and Crossman 1973).

Age at First Reproduction.— Males generally mature at about 28 cm (2 to 4 years) and females at 35 to 43 cm (3 to 6 years, Scott and Crossman 1973).

Life Span.— Maximum age varies from 10 to 12 years in the south, to 20 years in the north (Scott and Crossman 1973).

Body Size.— Growth is related to latitude, with individuals in southern populations tending to grow faster than those in the north (Scott and Crossman 1973; Quist et al. 2003). Walleye in the range of 5.4 to 8.6 kg regularly win angler contests in Ontario, but the largest individual on record from Ontario was just over 1 m in length, and 10.7 kg (Scott and Crossman 1973). Females are bigger than males at most age classes (Wolfert 1977).

Fecundity.— Fecundity is strongly related to age and has been given as high as 612,000 in females 800 mm in length (Scott and Crossman 1973). Fecundity-age relationships can be highly variable among populations, and may be partially related to food availability (Serns 1982; Baccante and Reid 1988). Fecundity did not vary with population density in one study in Wisconsin, and ranged from 41,061 to 53,009 eggs for female walleye in the 43 cm size class (Schueller et al. 2005). Fecundity relationships are given in Serns (1982), Baccante and Reid (1988) and Schueller et al. (2005).

6.18.5 Human Interest

The walleye is one of the most economically valuable freshwater fish species in Canada, as both a commercial and recreational target species (Scott and Crossman 1973; Wooding 1994). It is a major trophy fish in much of its range, and there are angling associations devoted solely to promotion and protection of this species. It is an important species in both summer and winter sport fishing, and can be caught by still fishing or trolling methods with live bait or lures (Scott and Crossman 1973; Wooding 1994). Walleye have been introduced in some systems as a "biological control" to increase predation on stunted fish populations (Galinat et al. 2002; Schneider and Lockwood 2002).

6.18.6 Distribution

Native Distribution.— The walleye is native to freshwater (and occasionally brackish water) across a broad range in central North America, east of the continental divide. Its northern limit extends from the Mackenzie River drainage near the Arctic coast, southeast to Hudson and James Bay, to the northern shore of the St. Lawrence River (Scott and Crossman 1973). It is present throughout or almost throughout Alberta, Saskatchewan, Manitoba, Ontario, and most of southern Quebec. It is naturally present in British Columbia in the Liard and Peace systems (McPhail and Carveth 1992). It is absent from the Maritimes, Newfoundland and Labrador, and most of the New England states; it is also absent from Alaska, California and Florida. It is present throughout the Great Lakes drainage and south to Missouri, Alabama and Georgia (Scott and Crossman 1973; NatureServe 2005).

Present Distribution.— Walleye continue to exist only in Canada and the US, but have been intentionally spread well-beyond their native distribution. It has been successfully introduced to the New England and Mid-Atlantic states, as well as all states west of its natural range, with the exception of California, where repeated introductions apparently failed (Fuller 2005; NatureServe 2005). It has been introduced in various places in Canada within its natural range, where barriers naturally prevented dispersal (Scott and Crossman 1973), and has spread via natural dispersal following intentional introductions.

BC Distribution.— The walleye naturally occurs in the Peace and Liard river systems, including portions within British Columbia (McPhail and Carveth 1992). After being introduced to the Columbia system in the US, the walleye has dispersed into the Columbia and some of its tributaries in southern British Columbia. In the Okanagan region, walleye are only currently known in Christina Lake, where a small reproducing population exists, but they might have originated from the Kettle River system in the US where they occur in a reservoir just south of the border (B. Jantz, pers. comm.). Thus, there is a possibility that they occur downstream in the Kettle River as well.

The following table summarizes unique waterbody occurrences within Ministry of Environment regional boundaries, based on records in the occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Region	No. of waterbodies
1	-
2	-
3	-
4	1
5	-
6	1
7	-
8	3
9	25

Figure 29 shows both the assumed native range of walleye in BC, as well as all individual occurrences in the province. The vast majority of observations are within the native range, with one observation in the northwest of the province (Skeena Region), and several more in the Southern Interior (Okanagan and Kootenay Regions). Additional waterbodies may be occupied by walleye, but these are not recorded in the occurrences database.

Stocking Records for BC.— There is only a single record in the releases database of authorized stocking of walleye in BC. 671 adult walleye were stocked in Cameron Lakes in Region 9, in 1988. However, walleye from Charlie Lake were apparently transplanted in Swan Lake in Region 9 by regional staff during the same period that yellow perch were moved from Swan Lake to Charlie Lake (in 1981).

Distribution Trends.— With only a few recorded observations outside the native range of walleye, the spread of this species is currently minimal, at least as indicated by the occurrences database.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
-	1	5	7

Mode of Invasion.— The walleye was intentionally stocked (authorized and illegal introductions) in the US as a food fish and for sportfishing (Fuller 2005). These fish have spread through natural dispersal, both upstream and downstream from release points (McMahon and Bennett 1996). It is likely that its presence in the Columbia and Okanagan watersheds are in part due to such upstream dispersal. Illegal introductions seem to be a growing problem in the western US (McMahon and Bennett 1996). Annual stocking supplements natural recruitment in many populations in the western US, and there is pressure to introduce this species to additional non-native waters in the Pacific Northwest (McMahon and Bennett 1996). It is not clear if illegal introductions have contributed to the spread of this species in BC.

6.18.7 Threats

As a top-level predator, walleye have the capacity to influence native fish communities and aquatic ecosystems, although these effects are generally less well-studied than those of other piscivores such as bass. This may simply reflect the fact that walleye have not been translocated to as many locations as bass. It seems likely that walleye will often have ecological effects similar to those of bass (see Section 6.13 and 6.16), including food web effects, changes in prey fish abundance, and possible local extirpations.

Following walleye introductions, fisheries scientists have frequently observed prey depletions, reduced salmonid populations, and large shifts in fish community composition (Colby and Hunter 1989 cited in McMahon and Bennett 1996). Predation by walleye has been shown to be a strong influence on many fish populations (e.g., Forney 1974; Lyons and Magnuson 1987; Galinat et al. 2002; Schneider and Lockwood 2002; Baldwin et al. 2003). Introduction of walleye to Seminoe Reservoir, Wyoming, led to increased costs for trout stocking because trout must now be released at larger sizes to avoid walleye predation; in other systems, trout stocking has been abandoned altogether (McMahon and Bennett 1996). Native minnow (*Hybognathus* spp.), darter (*Etheostoma* spp.) and sucker (*Catostomus* spp.) species, as well as crayfish (*Orconectes obscurus*), also declined sharply in this reservoir after walleye were stocked. Some systems have not been plagued by these difficulties, though the reasons are unclear (McMahon and Bennett 1996). Walleye are estimated to consume 2 million salmon smolts annually in the Columbia River, approximately one-third of total predation loss (McMahon and Bennett 1996). Walleye and salmonids appear to limit interactions in their native eastern habitat but western reservoirs generally lack sharp thermoclines and do not have abundant littoral habitat to allow separation (Fuller et al. 1999 and references cited therein). Frequently, non-native walleye deplete the forage base and become stunted, providing a poor fishery (McMahon and Bennett 19960. Management of walleye in the Columbia and elsewhere in the Pacific Northwest is now a difficult issue as managers must attempt to balance angler desires, biodiversity values, and multi-species trade-offs (Oregon Department of Fish and Wildlife 2005).

6.18.8 Potential Management Actions

Extirpating a population of walleye is likely to be difficult in most lake systems, and likely almost impossible in most large streams. Control would require the coincident use of several of the methods discussed in Section 5. Where other desirable fish species occur, it is essentially impossible to extirpate a single species once it becomes established. Efforts are thus best spent on preventing introductions. These efforts will require regulation, enforcement, education and stewardship. In theory, adequate statutes and regulations already exist: the possession, transport and introduction of any live fish into natural waters is prohibited without a permit (see Section 2.2). Unfortunately, these regulations are not stringently enforced, and compliance is insufficient to prevent spread of the species. Educational material should be developed and targeted at individuals and groups that are likely to translocate these (or other) fish. Strategies should be developed to ascertain who is likely to introduce fish, and to target communication at such groups and individuals. Coalitions with angler groups have had some success in the US, and opportunities for similar cooperation should be explored here. Strategies should also be developed that would aid public stewardship of freshwater bodies in BC, to prevent the introduction and spread of aquatic species in BC. Successful walleye fisheries in neighbouring states may tempt individuals in BC to promote similar fisheries in some parts of BC.

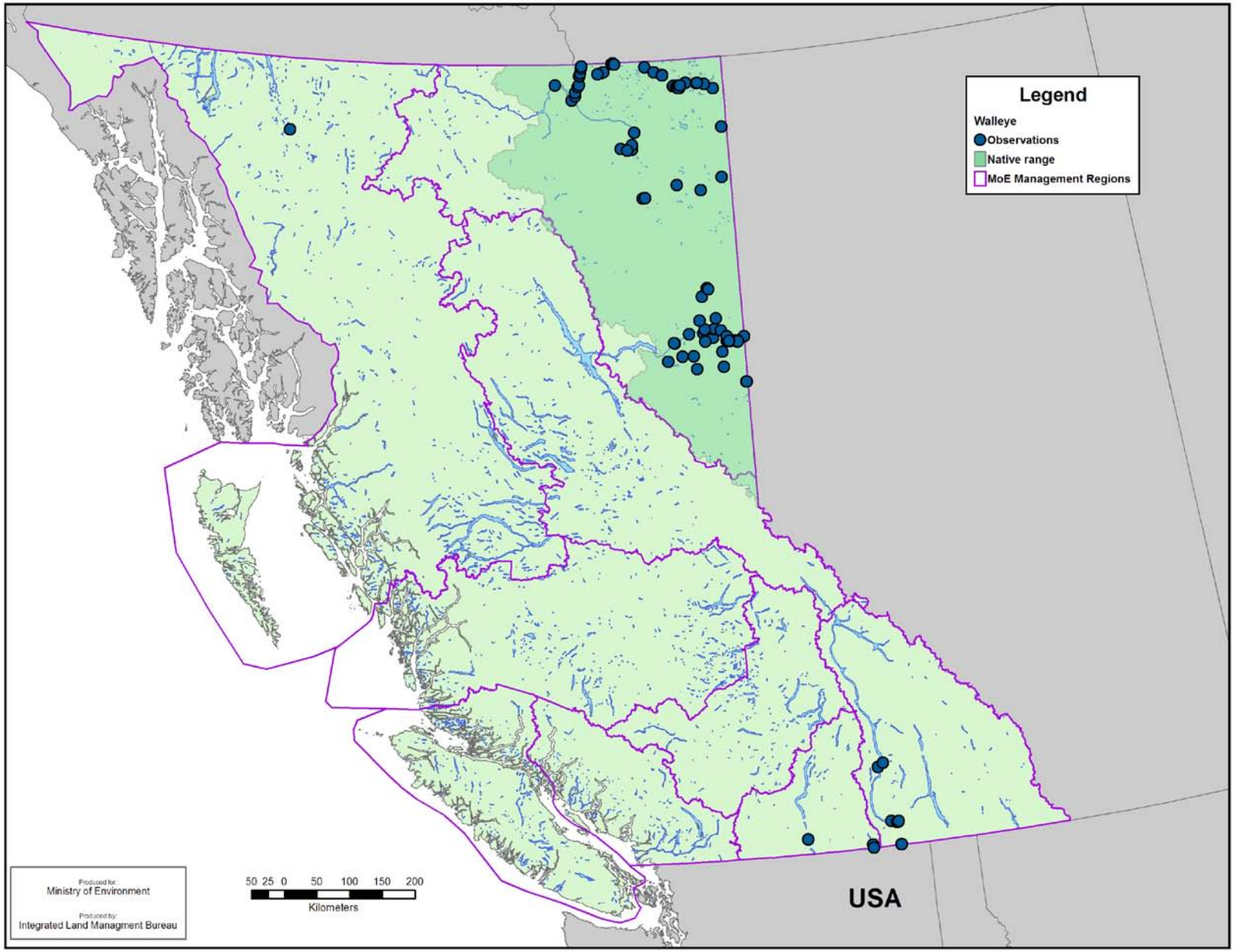


Figure 29. Walleye occurrences in British Columbia, based on the MoE occurrences database, and the species' assumed native range

Yellow Perch (*Perca flavescens*)[9]



6.18.9 Taxonomy

Family: Percidae (Perches)
Genus: Perca
Species: flavescens
Authority: Mitchill, 1814

Note: Some taxonomists refer to the yellow perch of North America and the perch of Eurasia as subspecies of *Perca fluviatilis*.

6.18.10 Habitat

Yellow perch are considered a shallow-water species, and are usually present in the upper 9 m of the water column (Scott and Crossman 1973). They are found in large lakes, small lakes and impoundments, as well as slow moving waters of large rivers, selecting areas with moderate vegetation and a sandy or gravel bottom (Idaho Dept. of Natural Resources 2005). This species has a high tolerance for low oxygen levels, but prefers moderate temperatures in the summer avoiding the deeper cold water or warm surface (Creque 2000). Adults and young aggregate in loose schools of 50 to 100 individuals, segregated by size (Scott and Crossman 1973). Young yellow perch tend to occupy shallower, nearer shore areas than larger fish (Scott and Crossman 1973). They are inactive at night and rest on the bottom (Scott and Crossman 1973). Studies of seasonal movements indicate that they tend to follow the 20 ˚C isotherm, but also respond to distribution of food items (Scott and Crossman 1973).

Spawning Habitat.— Spawning occurs in the shallows of lakes and tributary streams (Scott and Crossman 1973). Spawning microhabitats are variable, usually near rooted vegetation, submerged brush or trees, but sometimes over sand or gravel (Scott and Crossman 1973). After hatching, larvae initially stay nearshore, but then become pelagic until fins fully develop (Creque 2000).

[9] Image used with permission. Copyright © Noel Burkhead & Virginia Dept of Game and Inland Fisheries (Fishes of Virginia)

6.18.11 Diet

The diet of yellow perch is related to body size. Early diets are dominated by zooplankton, but shift as individuals grow, to benthic prey items such as odonates, Ephemeroptera, molluscs, ostracods, chironomids, and small fish (Scott and Crossman 1973). Larger perch (> 150 mm) can be predominantly piscivorous, and may be cannibalistic (Scott and Crossman 1973).

6.18.12 Life History

Reproduction.— Yellow perch spawn in the spring in water temperatures of 6.7 to 12.2 ˚C, usually from mid-April to early May, but sometimes well after this date (Scott and Crossman 1973). No nest or parental care is provided. Adults migrate into the shallows of lakes and often into tributary streams to spawn (Scott and Crossman 1973). Spawning occurs during day and night (Creque 2000). Females are thought to be followed by several males while they lay eggs, which are extruded in a large gelatinous, semi-buoyant egg mass (Scott and Crossman 1973). Aeration of the eggs is achieved passively, by means of ambient water movement through and over the mass (Scott and Crossman 1973). Hatching occurs after 8 to 10 days, followed by a further 5 days of inactivity during yolk absorption (Scott and Crossman 1973).

Age at First Reproduction.— Sexual maturity is achieved at 3 years for males and 4 years for females (Scott and Crossman 1973).

Life Span.— Maximum life span is usually 9 or 10 years, but occasionally individuals exceed this age (Scott and Crossman 1973; FishBase 2005).

Body Size.— Mature yellow perch in Ontario are usually in the range of 115 to 285 g (Scott and Crossman 1973). Maximum size is reported as 50 cm and 1910 g (FishBase 2005).

Fecundity.— Egg number per female is related to body size, and has been reported at 36,600 to 109,000 for fish 131 to 254 mm (Scott and Crossman 1973). Spawning females lay eggs in masses, which are an average of 23,000 eggs (range 2,000 to 90,000; Scott and Crossman 1973).

6.18.13 Human Interest

The yellow perch is a commonly exploited sport and commercial fish throughout its range. However, the commercial fishery is dominated by catches in the Great Lakes (Scott and Crossman 1973). The flesh is said to be superb (Wooding 1994). Anglers can take yellow perch with still-fishing methods, year-round (Scott and Crossman 1973; Wooding 1994). Yellow perch are also used as live and cut bait for angling other species (Scott and Crossman 1973). Yellow perch are common study organisms.

6.18.14 Distribution

Native Distribution.— The yellow perch is widely distributed in North America, east of the continental divide, from Nova Scotia to Alberta (Scott and Crossman 1973). The species occurs in the Great Slave Lake drainage, above 60 ˚N, and from here extends eastward to southern James Bay (Scott and Crossman 1973). The natural southern limit is in the US, from Nebraska to South Carolina (NatureServe 2005).

Present Distribution.— The yellow perch has been widely translocated in North America. The yellow perch now exists in almost all continental US states, with the exception of Alaska, Louisiana, Arkansas, Kentucky and Tennessee (NatureServe 2005). There are no records of yellow perch introductions in countries other than Canada and the US, although the European perch (*Perca fluviatilis*), a similar congener, has been introduced to Australasia (FishBase 2005). Yellow perch occur in all reservoirs in the mid- and lower Columbia and Snake rivers (Wydoski and Whitney 2003). It is now the most abundant fish in Lake Washington (Warner pers. comm. cited in Wydoski and Whitney 2003), and is established in various waters throughout Washington State (Wydoski and Whitney 2003).

BC Distribution.— The following table summarizes observation of yellow perch unique waterbody occurrences within MoE regional boundaries, based on records in the occurrences database. A description of the database, its limitations, and methods of analysis are discussed in Section 6.1.

Region	No. of waterbodies
1	5
2	4
3	3
4	16
5	-
6	-
7	-
8	17
9	7

The yellow perch is not native to British Columbia, with one possible exception of a single population in Swan Lake, on the BC – Alberta border. Yellow perch introduced to the Columbia drainage in Washington have dispersed into the Pend Oreille, Columbia, Kootenay and Okanagan within BC. Yellow perch also have been introduced to a number of waterbodies in other areas including Vancouver Island, Lower Mainland and Kamloops regions.

Distribution Trends.— Although the total number of observations is still fairly low, data indicate a steady spread of yellow perch in BC. Occurrences are concentrated in fairly limited areas (mostly southern BC and eastern Peace region). Most observations have occurred in the last 30 years, although many records are not dated.

Total no. of waterbodies up to 1950	Total no. of waterbodies up to 1975	Total no. of waterbodies up to 2005	Total in database
-	10	31	51

Stocking Records for BC.— There are no specific records in the releases database of authorized stocking of yellow perch in British Columbia. However, regional files indicate that a number of lakes in the Peace region were stocked once with yellow perch during the 1980's. These

introductions were conducted by provincial government staff. Bearhole, Radar, Stony and Kelly lakes were all stocked with fish from Sturgeon Lake in Alberta, while Charlie Lake was stocked with fish from Swan Lake. Of these lakes, it is believed that populations were only established in Bearhole, Stony and Charlie lakes. Finally, there is no documentation of introductions in Swan Lake where an established population now occurs. However, given its isolation from any other native population and the fact that no perch were recorded in historic surveys, it is highly likely that this lake was originally stocked with yellow perch, possibly during the 1970's from an Alberta source (B. Anderson, pers. comm.). Authorized introductions in the Peace region are listed in the following table.

Waterbody	MoE Region	Year
Bearhole Lake	9	1984
Radar Lake	9	1984
Stony Lake	9	1984
Kelly Lake	9	1984
Charlie Lake	9	1981

Mode of Invasion.— Historically, yellow perch have been introduced to non-native waters to enhance recreational and commercial fishing opportunities. Most introductions have been successful (Fuller 2005f). It is likely that the species has spread beyond points of introduction, via natural dispersal. In BC, the only authorized stocking of yellow perch occurred in the Peace region where yellow perch were introduced into five lakes during the early 1980's. The remaining occurrences are due to illegal introductions and subsequent spread. Periodic occurrences of yellow perch in the Peace River mainstem might be naturally derived from downstream populations in Alberta, or from dispersal from nearby Charlie and Swan lakes (B. Anderson, pers. comm.). Natural dispersal northward from the US is a possibility in several cases. It is thought that illegal stocking is carried out by individuals with the intention of providing a forage base for salmonids or bass species. Regional biologists have noted a pattern of introductions in some small lakes; first, yellow perch show up, and then bass follow. Given the flexible habitat preferences of yellow perch, the potential for spread in BC is extensive. One of the largest concerns currently is that the yellow perch found in small lakes in the Shuswap Lake region may move downstream into the large lake and anadromous river sections where valuable native trout and salmon populations reside (R. Bison, pers. comm.).

6.18.15 Threats

There have been significant concerns expressed by some at the human-aided range expansion of yellow perch that is occurring in BC. Wooding (1994) notes that concerns about the spread of yellow perch exist because they have a high reproductive capacity and are predators of eggs and juveniles of trout (and presumably other species). They also compete with trout for food (Coots 1966 cited in Fuller et al. 1999). Fisheries managers and culturists in BC have observed that wherever yellow perch have been introduced in small lakes, subsequent stocking and survival of small hatchery salmonids is almost completely unsuccessful (T. Yesaki, pers. comm.). In their native range, yellow perch are an important food source for top predators including walleye, northern pike and lake trout in colder waters (Creque 2000). In BC lakes without a similar predator analogue, yellow perch abundance can be very high.

Despite the stated concerns we found little reliable information on yellow perch as a non-native species. For example, a search of multiple web-based databases on invasive species produced no results for yellow perch as a non-native species with ecological impacts. We also found no information in the published scientific literature on the effects of yellow perch as a non-native species.

6.18.16 Potential Management Actions

Most fish species are difficult to eradicate once established. Yellow perch may be especially difficult given their early maturation, high fecundity and fast growth rates. Closs et al. (2003) noted that physical removal of Eurasian perch (a closely-related species) was feasible in some instances in New Zealand, but in others removal actually increased recruitment, presumably due to the removal of cannibalistic adults. Efforts are probably best-spent on trying to prevent human-aided dispersal of this species into new habitats. Information on yellow perch should be included in material used to educate the public on the ecological dangers of translocating wildlife. Regulations exist prohibiting the transportation or introduction of live fish without a permit. These regulations are likely useful, though they remain difficult to enforce, and will therefore be an incomplete defence against further spread of this species.

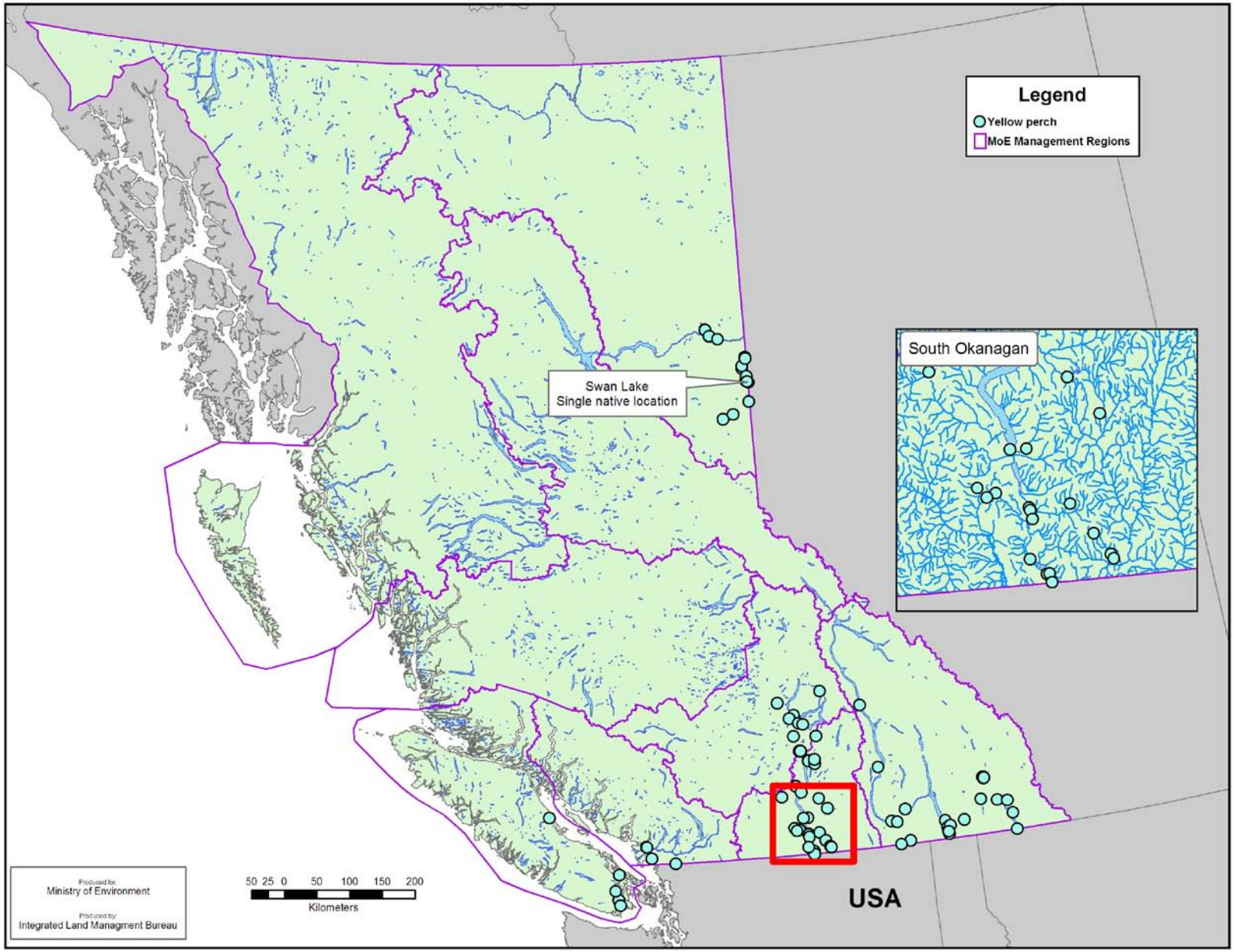


Figure 30. Yellow perch occurrences in British Columbia, based on the MoE occurrences database.

7. PROVINCEWIDE PATTERNS AND TRENDS

British Columbia has experienced a significant increase in the presence of non-native freshwater fish species, both in numbers of systems and in numbers of species. Data indicate a growing number of non-native freshwater fish species in BC Table 4. Data also indicate a growing number of waterbodies with one or more non-native freshwater fish species Table 5. Some of this increase can be explained by the increase in the number of watersheds that have been surveyed in the province, but it seems likely that these patterns represent a true trend in the spread of non-native fish species within BC. Regions with the greatest increase in distribution of non-native species are those with the greatest human population in the province: Vancouver Island, Lower Mainland, Okanagan and Kootenays. The general trend of increased distribution of non-native species is likely even more pronounced than that indicated in Table 4 and Table 5, because this comparison does not include species native to BC that have been introduced into waters outside of their natural range (with the exception of walleye, northern pike and yellow perch, all of which have very limited native ranges in BC).

Table 4. Number of non-native freshwater fish species in BC in three time periods. Data source: provincial freshwater fish occurrence database.

No. of species up to 1950	No. of species up to 1975	No. of species up to 2005
7	13	18

Table 5. Summary of unique waterbodies in each MoE region with one or more non-native freshwater fish species. Data source: provincial freshwater fish occurrence database.

Region	No. of waterbodies up to 1950	No. of waterbodies up to 1975	No. of waterbodies up to 2005	Total in database
1	4	37	84	104
2	4	28	57	93
3	2	13	40	61
4	10	63	325	398
5	1	12	17	20
6	1	3	6	11
7	1	4	19	19
8	4	34	73	139
9	1	1	4	15

In terms of distribution of non-native species in BC, two patterns are obvious. The first is that observations tend to be focussed in and around urban areas. Secondly, introductions are more concentrated in the southern regions of the provinces, particularly in watersheds close to the border or that share waters with the US.

The concentration of non-native species in and around urban areas is not surprising. This pattern is observed both in species whose main route of introduction is via authorized stockings

(e.g., brook trout and brown trout) and for those whose presence is predominantly the result of illegal introductions (e.g., bass species, yellow perch, and catfish). Small lakes and streams near urban areas have been stocked to enhance angling opportunities that are close to cities and towns. In the case of small lakes, this stocking initiative continues. With respect to unauthorized introductions, proximity to urban areas increases the likelihood that an unwanted pet may be released, that organisms from man-made ponds escape, or that individuals who would like to create their own fisheries close to home may be moving fish around. In addition to these reasons, biological survey effort is likely greater in and around urban areas due to easier access and a greater need to monitor or inventory these watersheds over the years.

The concentration of non-native species in southern areas of British Columbia may reflect both environmental conditions and proximity to US border, although this portion of the province is also the most densely populated. Climate tends to be milder in the south providing conditions more hospitable to warm-water species such as bass, carp and bullheads. Greater levels of disturbance in this region may also contribute to the spread of non-native species. Some of the greatest changes to habitat have occurred in the Columbia and Kootenay watersheds where a series of major hydro-electric and flood-control dams have fragmented and altered habitat. Elsewhere, the conversion of natural riverine habitat to lentic habitat found in reservoirs has been accompanied by the replacement of many species by non-native species better adapted to the new environment (Quist et al. 2005). Forestry-related activities can increase water temperature in streams; water temperature may play a key role in the likelihood of displacement of coldwater species (e.g. cutthroat trout and bull trout) by species with less stringent requirements (e.g., brown and brook trout) (Rieman et al. 2006) . While forestry occurs throughout most of BC, the effect on non-native species' distributions may be more relevant in southern watersheds where the presence of non-native species is already greater. Lastly, proximity to the US border increases the likelihood of both illegal introductions from easily accessed non-native populations in the US, and natural northward dispersal in shared watersheds. For example, the Columbia River in BC can receive migrants from downstream river sections in Idaho and Washington, and the Kootenay and Flathead rivers can receive migrants from upstream sections in Montana. In the past, waterbodies in northwestern US states have been stocked with walleye, bass species and yellow perch. Walleye in the Columbia system within BC likely originated from Washington where they were stocked in Lake Roosevelt (Fuller et al. 1999). The presence of brook trout in Ross Lake, a border lake in the Skagit watershed, is due to authorized introductions in Washington. The recent documentation of the green sunfish in Osoyoos Lake is likewise believed to be due to movement from Idaho waters (D. McPhail, pers. comm.).

7.1 Regional Trends and Patterns

The occurrences data are likely conservative for all species and in all regions. The regions vary not only in the number of occurrences but also which species are present. For example, most carp observations are concentrated in the south, black crappie and brown bullhead are concentrated almost entirely in the lower Fraser River area, and introduced walleye are noted only in the Okanagan and Columbia watersheds. The following describes general species distributions by MoE region.

Vancouver Island (Region 1): The predominant species found on Vancouver Island are Atlantic salmon, centrarchids (smallmouth bass and pumpkinseed sunfish), yellow perch, brown bullhead and brown trout (see Figure 34). With the exception of Atlantic salmon, which originate from aquaculture facilities, non-native species are concentrated in and around urban areas from Victoria to Campbell River, in watersheds most frequently accessed by the public. Only a few records are from locations north of Campbell River or on the west coast. The presence of most of these (with the exception of the salmonids), is likely due to illegal introductions by individuals or groups wishing to create a fishery. Brook trout were repeatedly introduced to southern Vancouver Island by the provincial hatchery program in the past (although stocking was terminated in 1990) (Figure 17). However, only one population is known to persist as a naturalized population (S. Rimmer, pers. comm.), suggesting that Vancouver Island does not provide appropriate habitat to support this species.

Lower Mainland (Region 2): The predominant species found in the Lower Mainland area are centrarchids (black crappie, largemouth bass, pumpkinseed sunfish), carp and brown bullhead (Figure 35), the majority of which have been introduced illegally with the intention of creating or enhancing a fishery. Brook trout stocking was limited in this region and was terminated in 1980.

Thompson (Region 3): The majority of observations in this region are brook trout associated with the provincial stocking program (Figure 36). More recently, the number of systems containing smallmouth bass or yellow perch has increased significantly suggesting that illegal movement is ongoing, or even increasing in effort. Carp observations have also been made in the South Thompson, as well as a single record of American shad and Atlantic salmon, both of which likely originated from salt water.

Kootenays (Region 4). The predominant species found in the Kootenays region is brook trout (Figure 37), which originates from the ongoing provincial stocking program. Other species with a significant number of observations include walleye, largemouth bass (and other centrarchids), and yellow perch. Each of these is associated with illegal movement and/or dispersal from south of the border. Common carp are also found in this region. The Kootenay region has by far the greatest number of non-native species occurrences (Figure 33), when species introduced by both authorized and unauthorized means are considered.

Cariboo (Region 5): This region is relatively pristine in comparison to the southern regions and the majority of observations are associated with stocked brook trout (Figure 38). A few Atlantic salmon observations in the coastal systems have been noted. The recent finding of smallmouth bass in a tributary of the Quesnel River suggests that this species may be more widespread here than thought.

Skeena (Region 6): This region is also relatively free of non-native species (Figure 39). Occurrences include mainly stocked brook trout. Bullhead occur in a couple of coastal watersheds, as do Atlantic salmon, which are likely escapees from aquaculture facilities.

Omineca (Region 7): This region has received only limited numbers of brook trout (currently and in the past), concentrated in the upper Fraser watershed. The only other records of non-native species are a single goldfish and brown catfish documentation (Figure 40).

Okanagan (Region 8): A large number of non-native species have been documented in this region (Figure 41). Brook trout are the predominant species numerically, due to authorized stocking programs both past and present. Walleye, yellow perch, brown catfish pumpkinseed sunfish, carp and smallmouth bass occur in systems close to or connected to the US border (i.e., Okanagan and Kettle watersheds).

Peace (Region 9): This region is relatively free of non-native species (Figure 42). There are a few brook trout occurrences, largely associated with past and present stocking programs. Yellow perch, a single record of fathead minnow and two carp records, are also in the database.

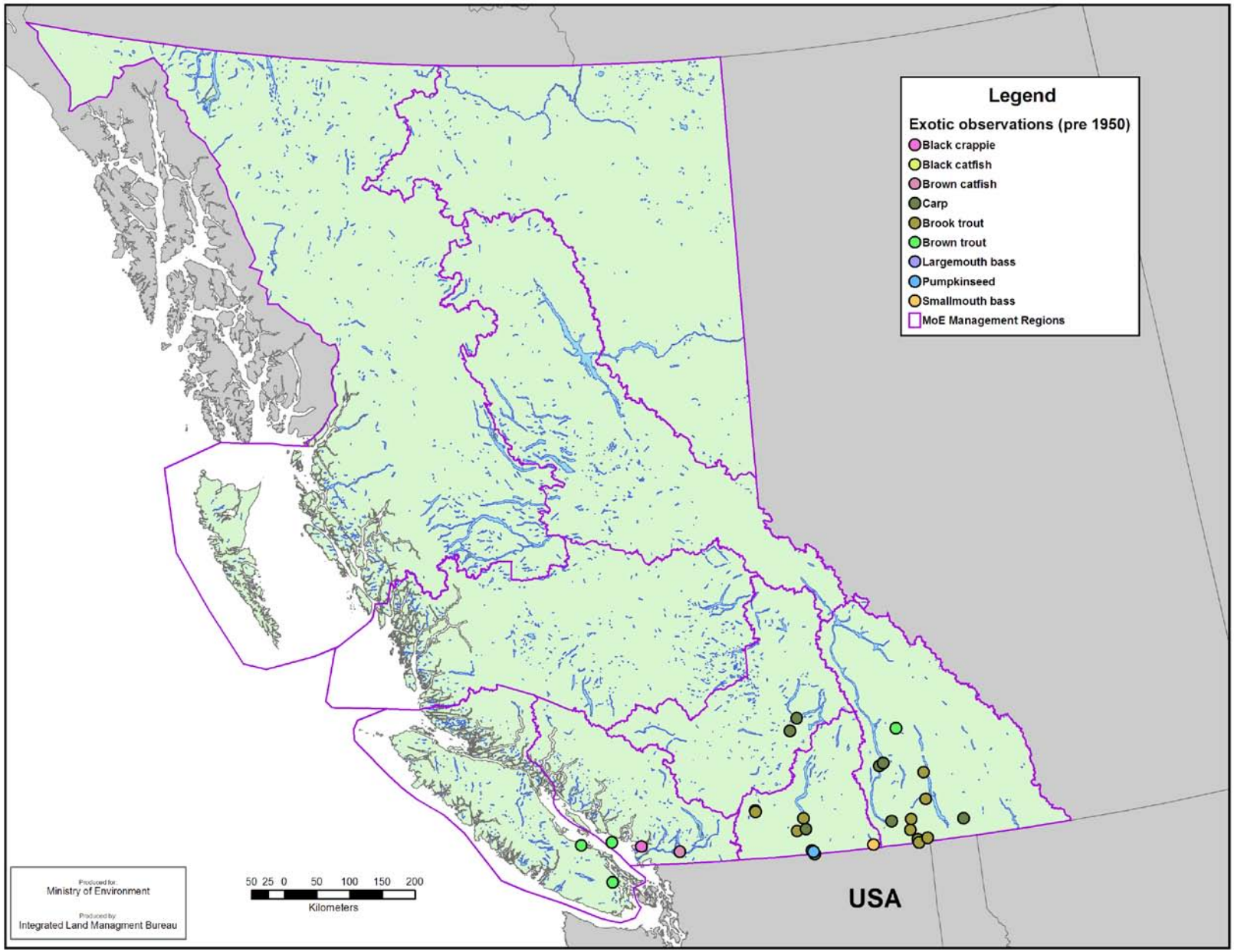


Figure 31. Observations of non-native fish species in British Columbia prior to 1950.

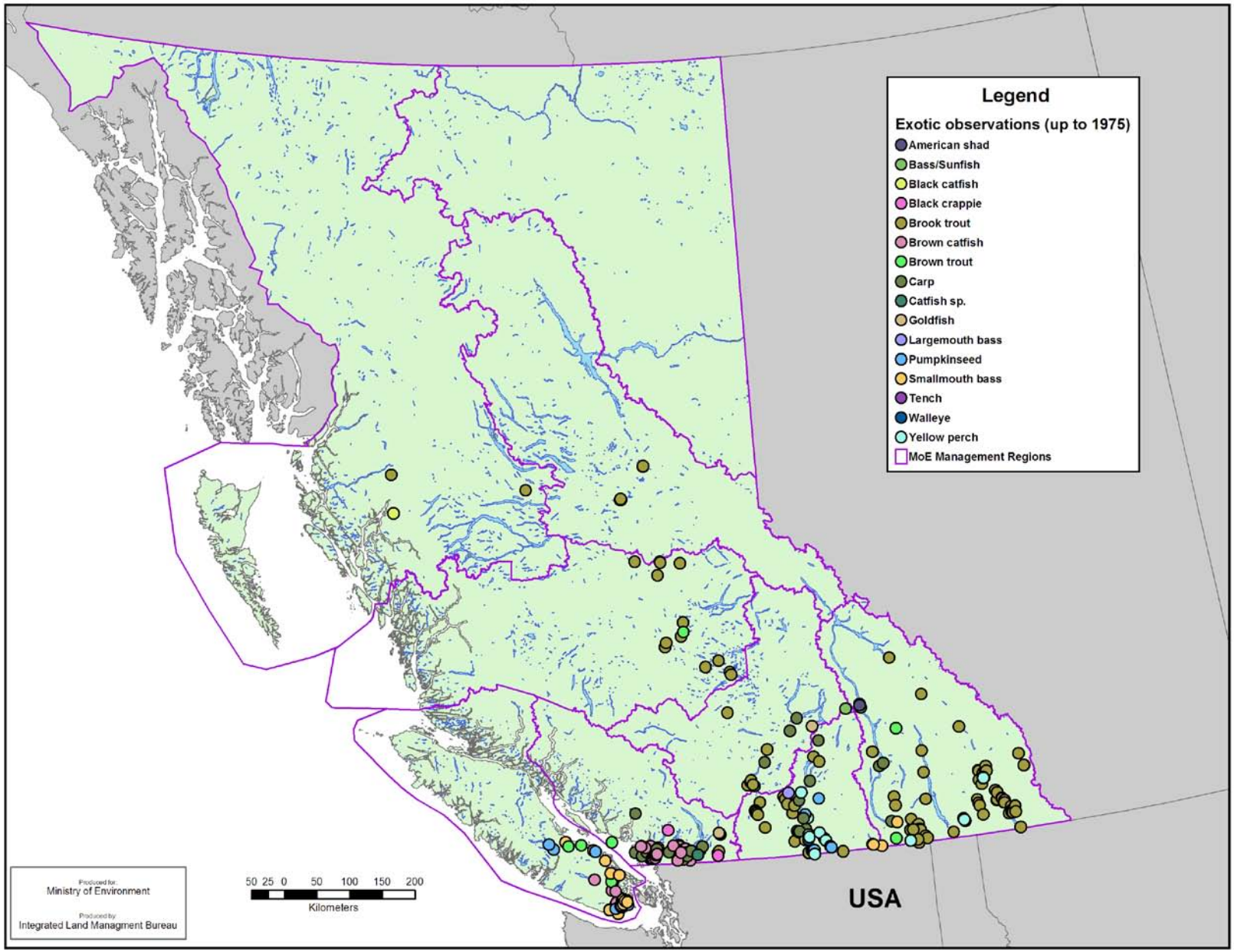


Figure 32. Observations of non-native fish species in British Columbia prior to 1976.

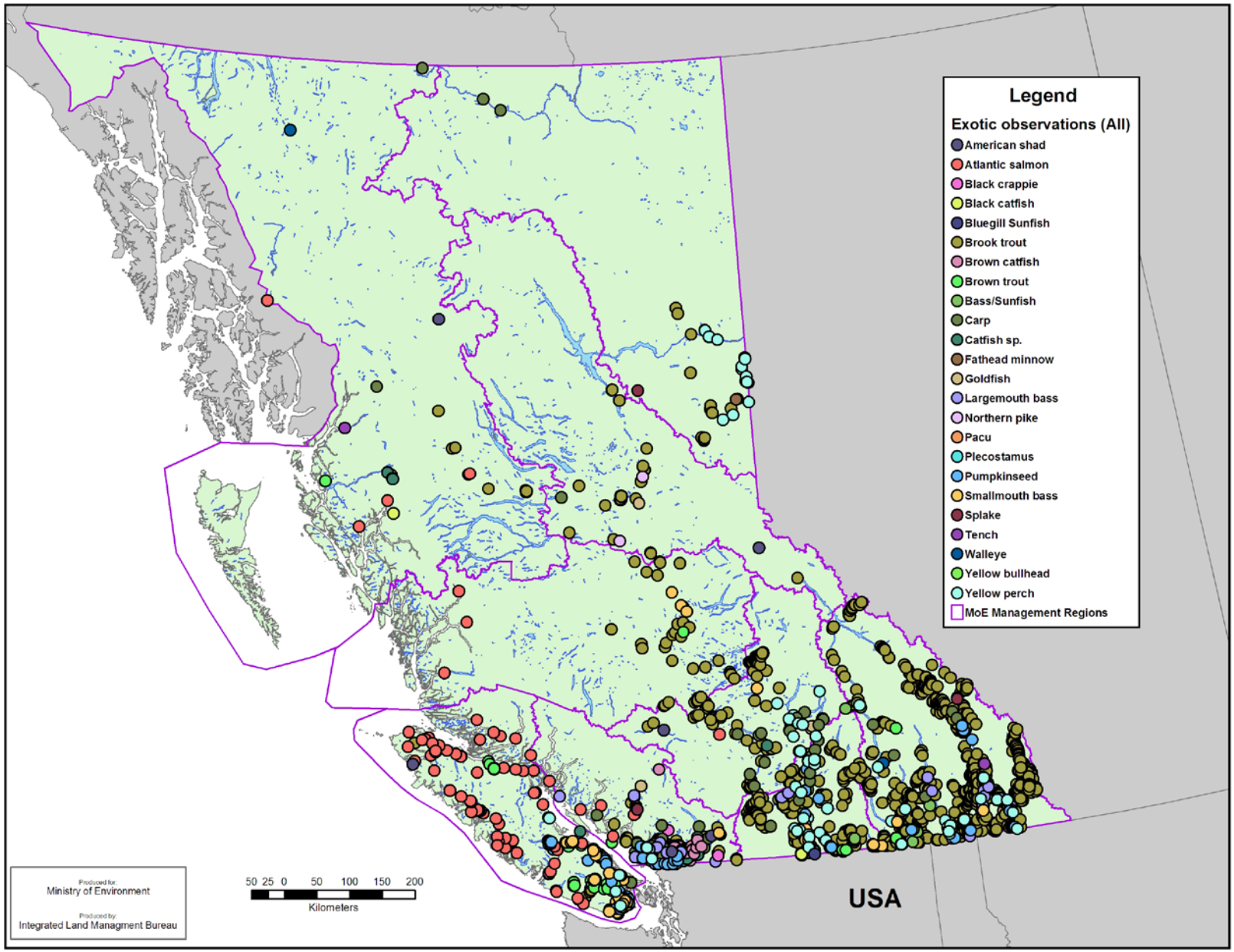


Figure 33. Observations of non-native fish species in British Columbia, based on all records in the MoE occurrences database.

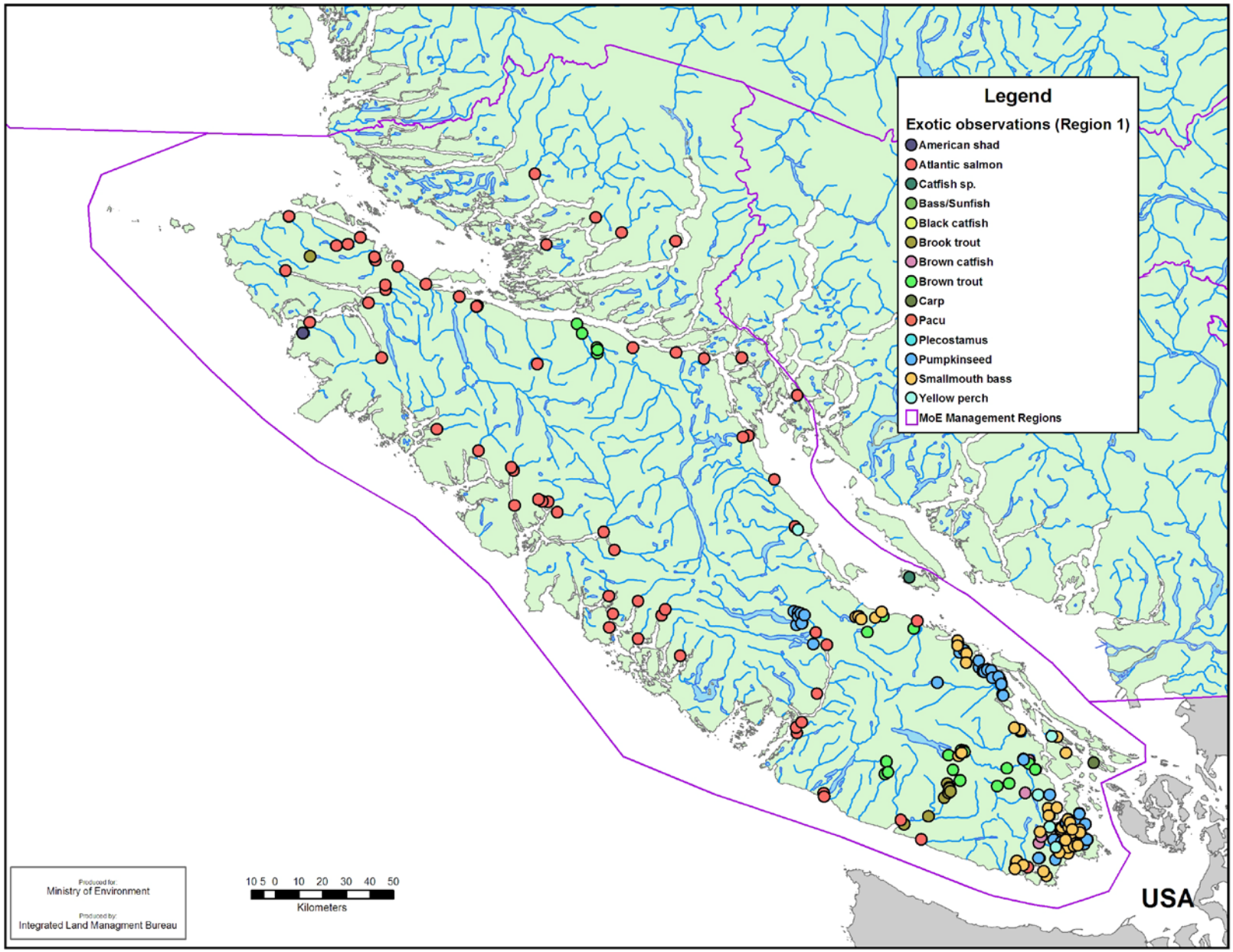


Figure 34. All observations of non-native fish species in MoE Region 1 (Vancouver Island), based on records in the MoE occurrences database.

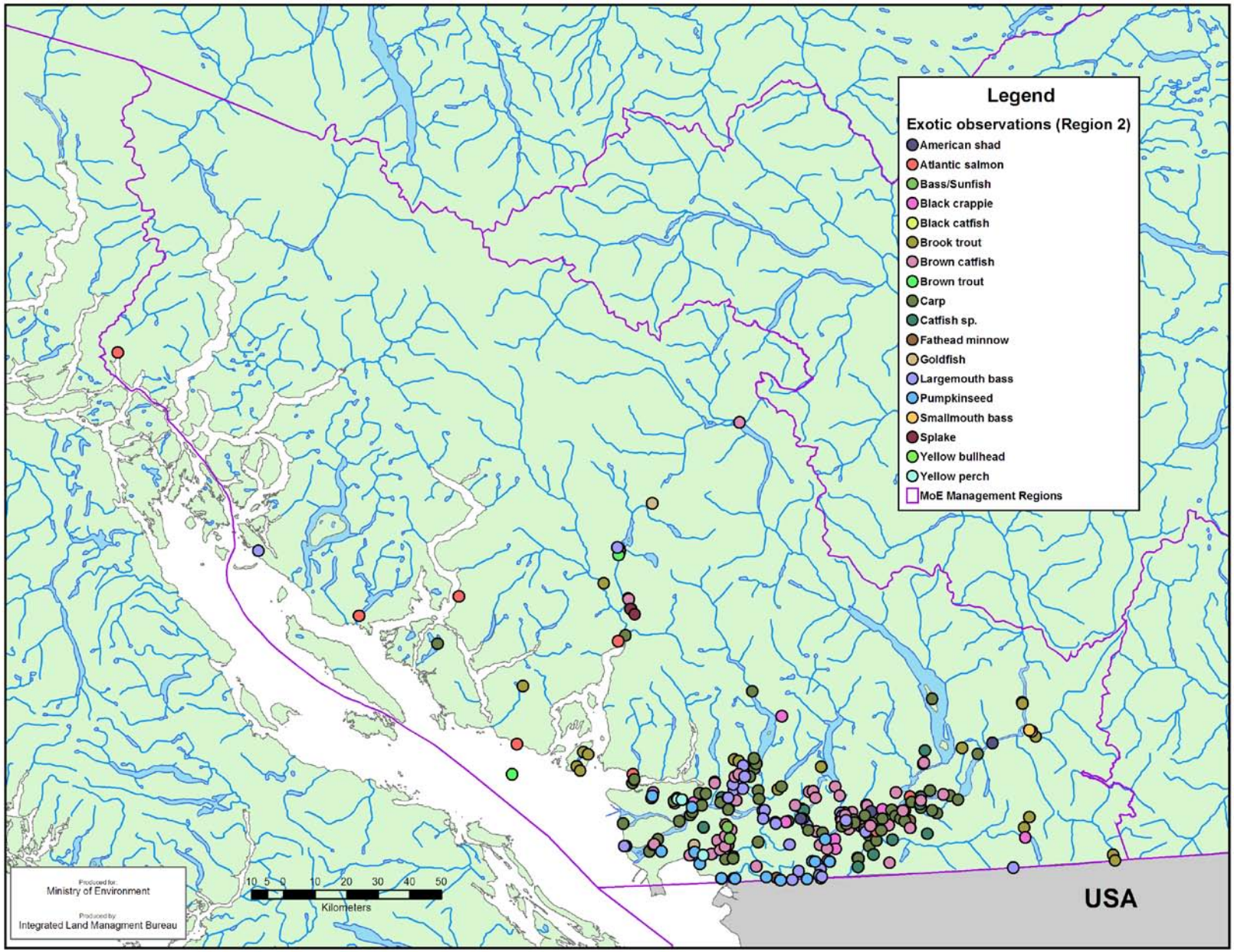


Figure 35. All observations of non-native fish species in MoE Region 2 (Lower Mainland), based on records in the MoE occurrences database.

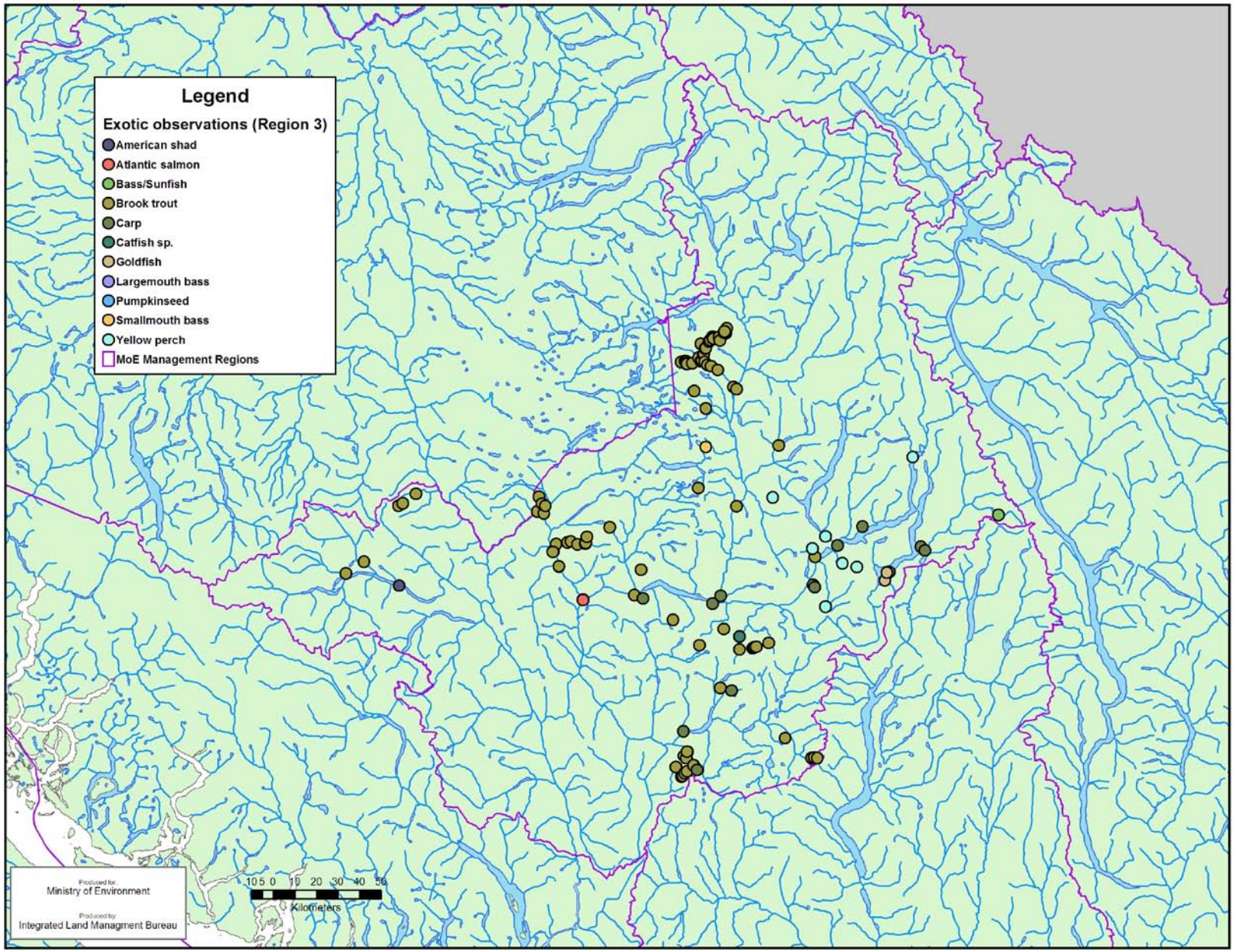


Figure 36. All observations of non-native fish species in MoE Region 3 (Thompson), based on records in the MoE occurrences database.

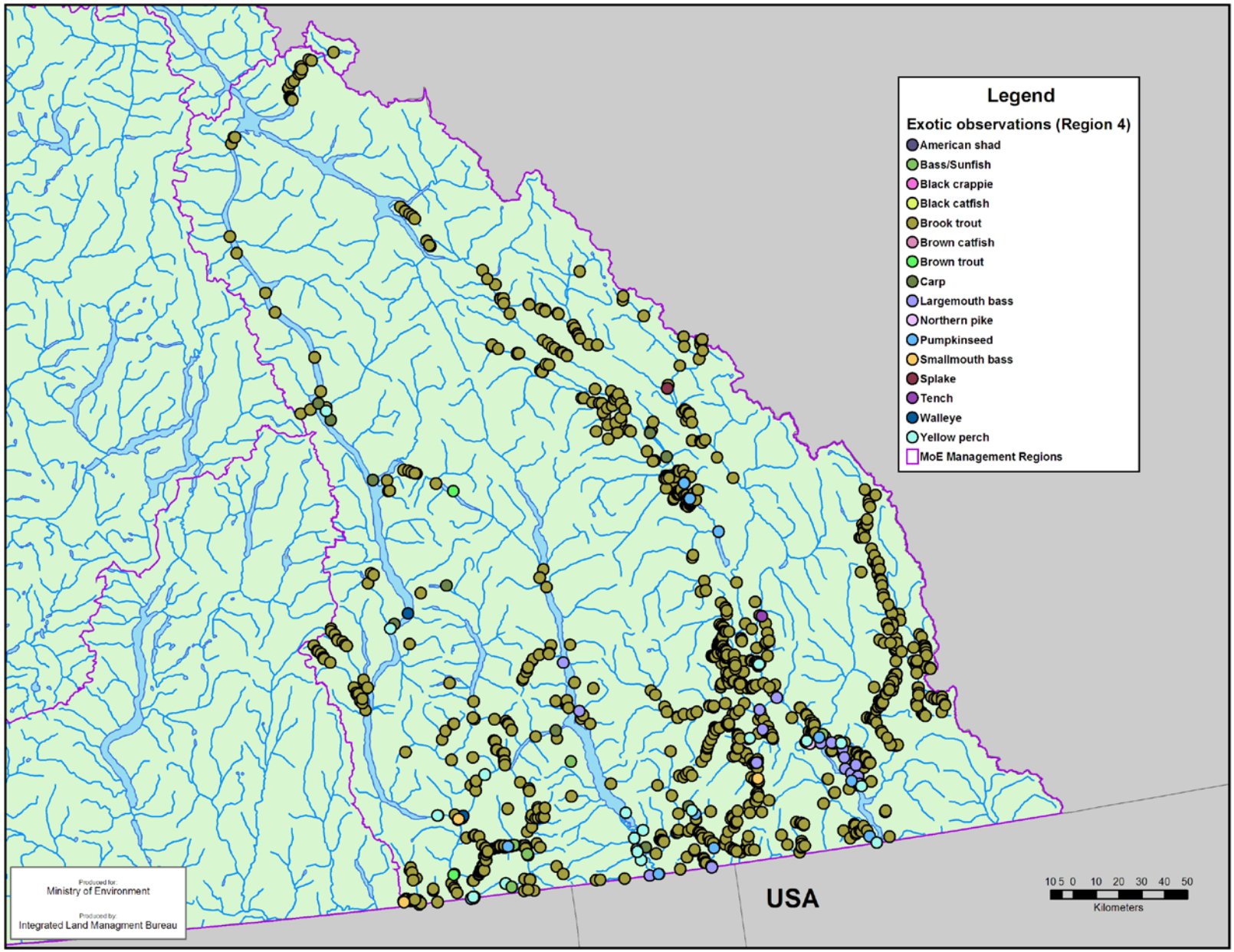


Figure 37. All observations of non-native fish species in MoE Region 4 (Kootenay), based on records in the MoE occurrences database.

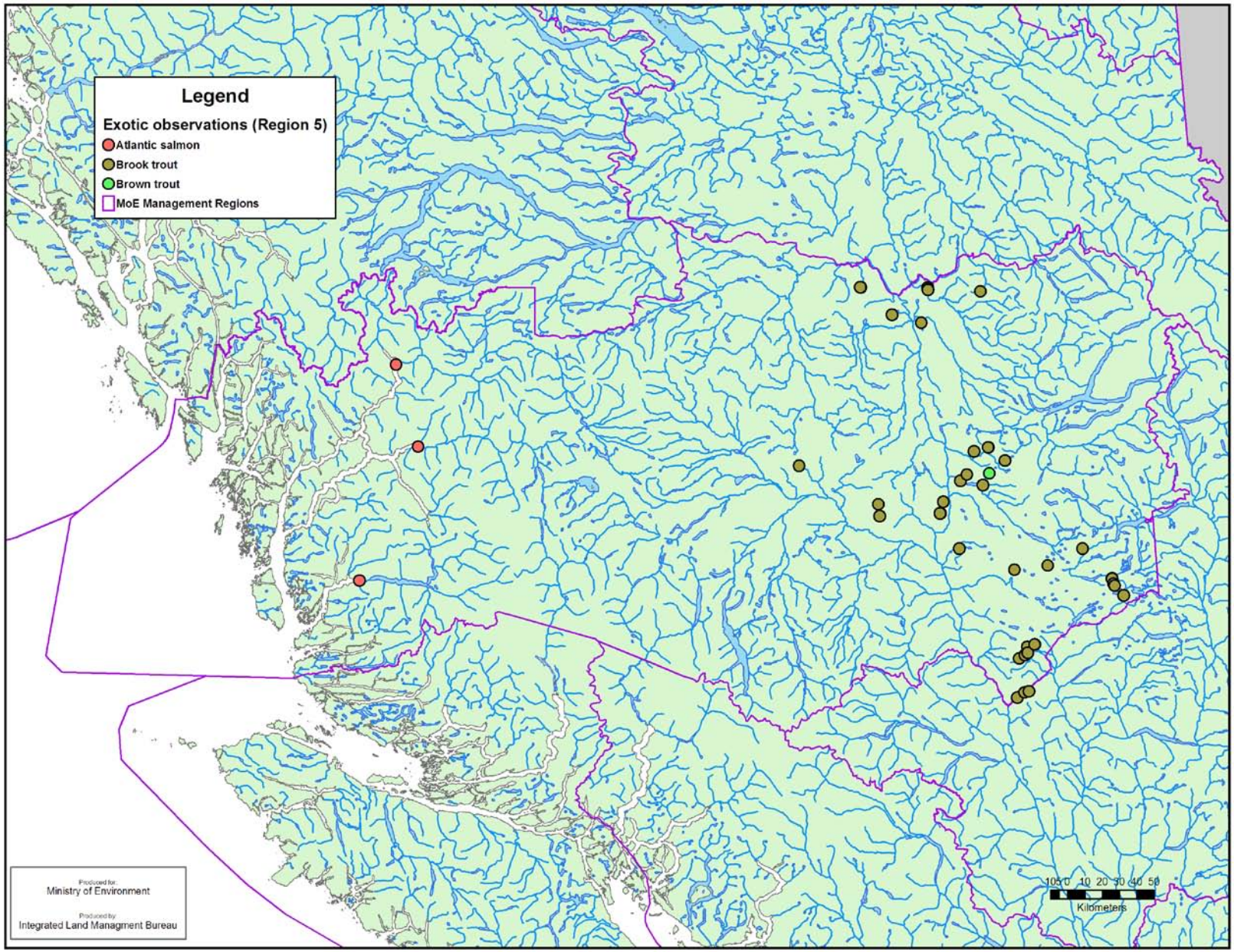


Figure 38. All observations of non-native fish species in MoE Region 5 (Cariboo), based on records in the MoE occurrences database.

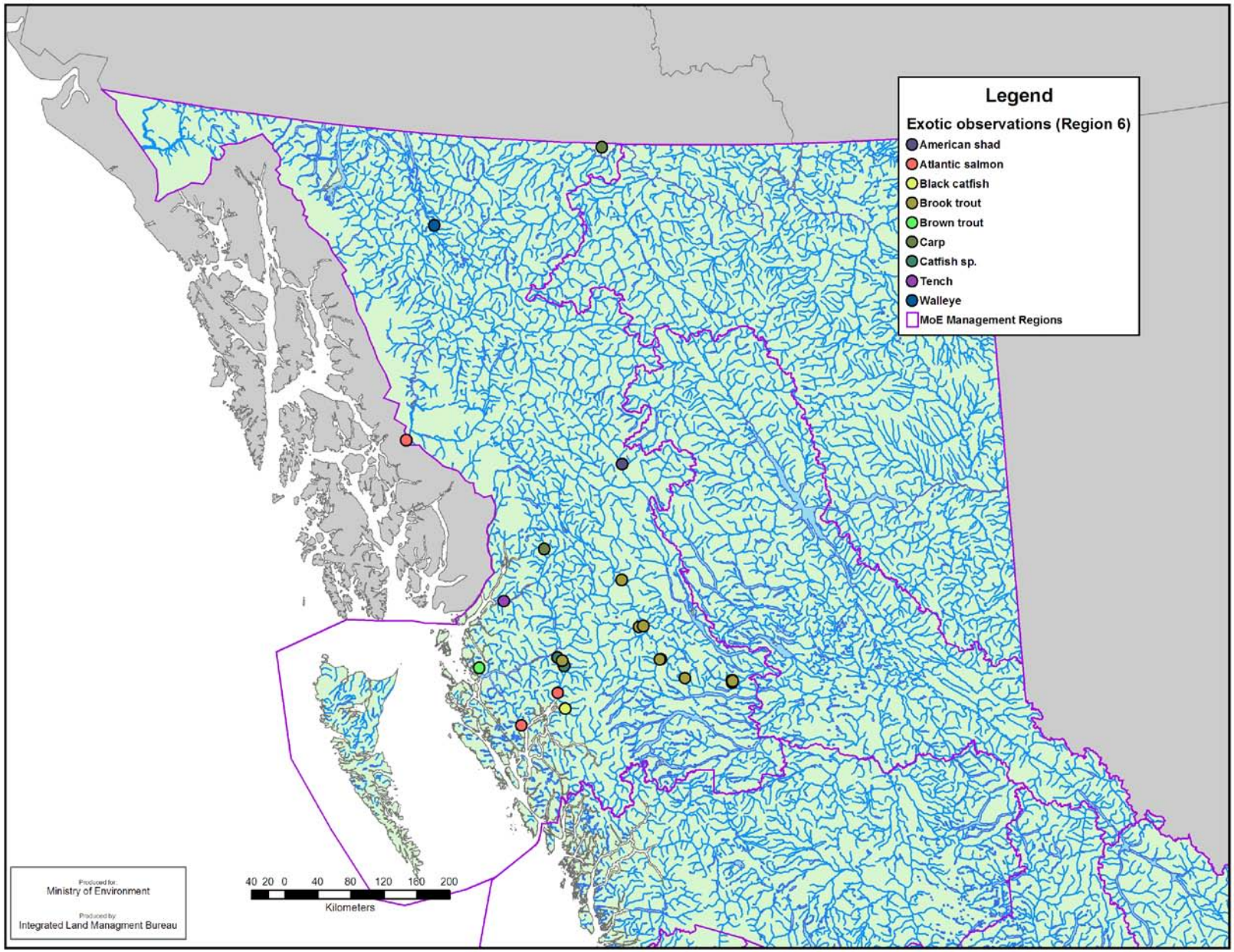


Figure 39. All observations of non-native fish species in MoE Region 6 (Skeena), based on records in the MoE occurrences database.

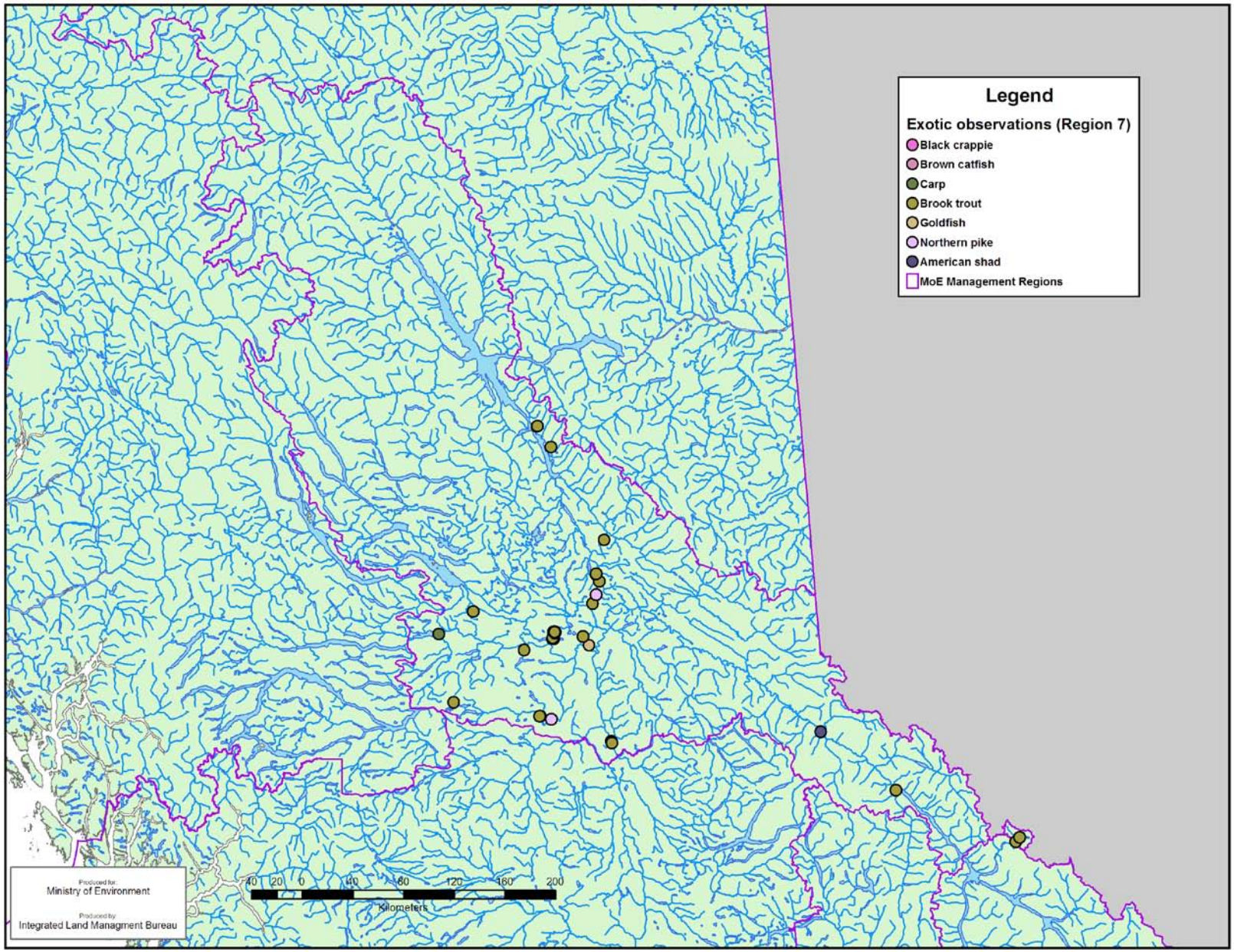


Figure 40. All observations of non-native fish species in MoE Region 7 (Williston), based on records in the MoE occurrences database.

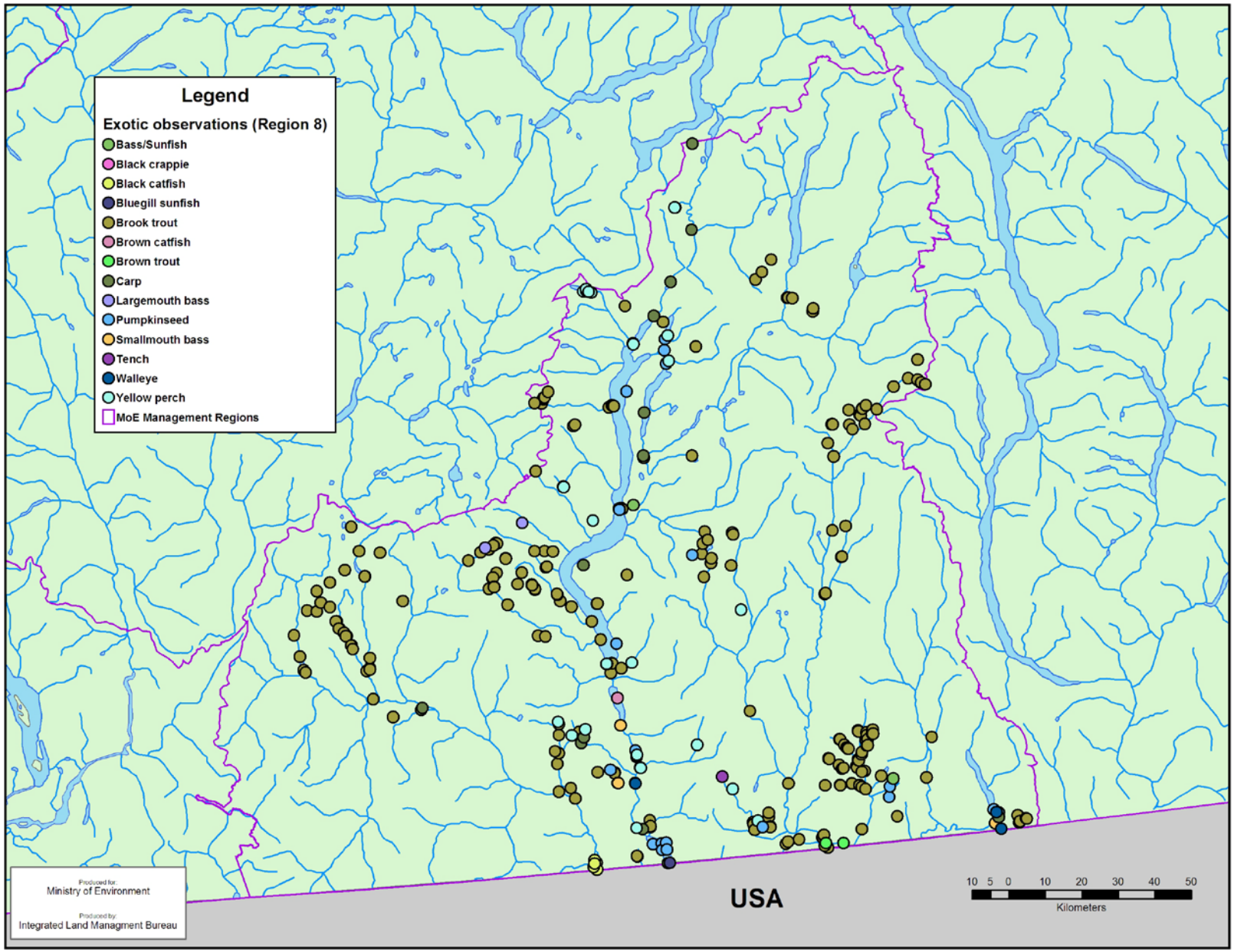


Figure 41. All observations of non-native fish species in MoE Region 8 (Okanagan), based on records in the MoE occurrences database.

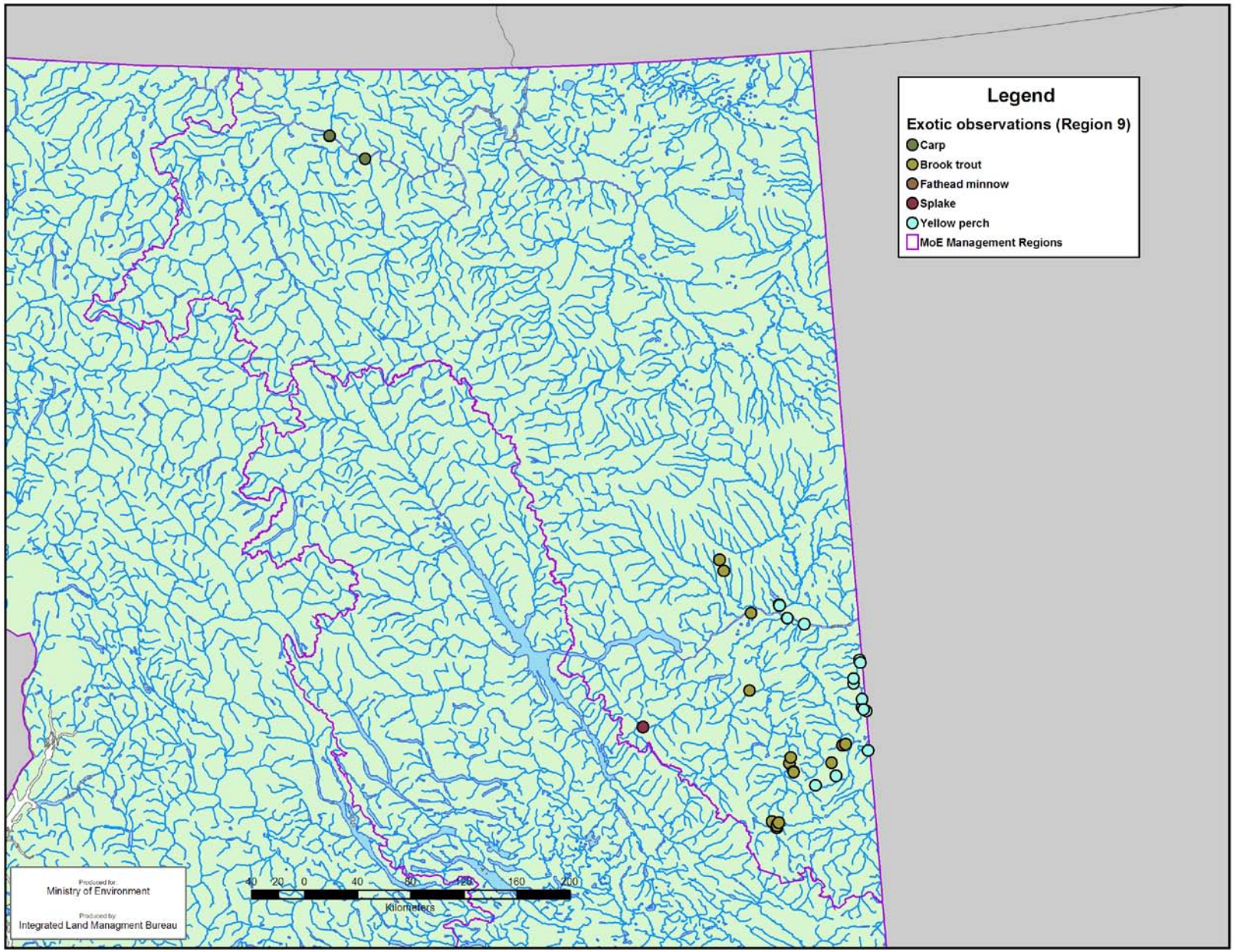


Figure 42. All observations of non-native fish species in MoE Region 9 (Peace), based on records in the MoE occurrences database.

7.2 Recommendations

The problem of potentially invasive fish is not equally distributed across British Columbia and different species identified in this document pose different threats to biodiversity. We must also recognize that probability of establishment or threats to biodiversity cannot be easily predicted for most species. Therefore, monitoring of non-native species distributions should be undertaken regularly. However, given limited resources there are certain geographic areas and species on which more intensive efforts should be focused. These priority areas can be grouped according to mode of introduction into BC, as follows.

7.2.1 Authorized Stocking Programs

Several salmonid species have a limited record of stocking (i.e., whitefish, lake trout, Arctic grayling), are stocked no longer and given their biology are unlikely to establish new populations. There are no recommended efforts to manage these introduced species at this time.

Cutthroat trout (westslope and coastal subspecies), brook trout and rainbow trout are still stocked extensively in small lakes, and to some extent in anadromous streams. Stocked fish are for the most part not from the receiving watershed (except for some anadromous programs), or in the case of brook trout not from BC. These stocking programs need to be reviewed and evaluated in light of the potential threats they pose to native species described in this document. There may be a number of stocking practices that can be altered to minimize these threats to biodiversity values, while maintaining most recreational benefits. For example, hatchery fish should not be stocked into small lakes where wild populations of the same species occur unless it can be demonstrated that hatchery fish pose little or no risk to their wild counterparts. Given that BC's fish culture program is mainly to support recreational opportunities rather than rebuild wild populations, sterile fish should be used as much as possible to minimize impacts to ecosystems. Regional stocking programs should consider phasing out the stocking of a species out of its native range (e.g. rainbow trout in the Upper Kootenay watershed). There are also several rearing and release strategies for anadromous programs that should be maintained to ensure impacts to native conspecifics are minimized; for the most part, provincial stocking programs have incorporated these strategies.

One additional stocking program is worth mentioning here. The historic brown trout stocking program focussed mainly in the Cowichan River was terminated in the 1980's. However, a naturalized population persists. Occasional reports of brown trout in other systems suggest that this resident population may be capable of limited saltwater movement. Given its status as one of the most invasive species globally, new reports of this species should be carefully verified to ensure it is not spreading beyond original points of introduction, or threatening existing native species in the Cowichan River. For example, inventory and stocking assessment crews should collect any voucher specimens of brown trout in systems outside of the Cowichan River.

7.2.2 Illegal/Unauthorized Introductions

The unauthorized movement and introduction of both native and non-native fish is not a recent phenomenon. Such movement can occur both intentionally and non-intentionally. However,

the presence of some non-native sportfish species (namely bass species, yellow perch and northern pike) in certain geographic areas appears to be fairly new. The patterns and species involved suggest very deliberate efforts specifically to create new fisheries. This is of great concern, threatening fisheries for native species, as well as native biodiversity values that extend well beyond the fish community. Once these species are established, they are impossible to eliminate except through extreme means such as piscicide application and even then only under very limited circumstances. Thus, the most effective means of eliminating or reducing the risks these species pose is prevention of illegal movement in the first place. In theory, existing statutes and regulations to prevent deliberate introductions should be adequate (see Section 2.2); however, enforcement is difficult and the spread of species to new, inaccessible locations indicates that compliance is clearly lacking.

Unauthorized introductions can only be prevented through strong public support and awareness. In particular, communities and individuals must be convinced that native species and native species fisheries are more valuable than non-native sportfish if prevention strategies are to be successful. Regulations could be improved to address importation and trade of non-native species. In particular, the Prohibited for Import Species list should be updated (e.g. add species such as all snakehead spp.) or an equivalent list developed under the BC Wildlife Act, and vendors (including food, pond, and aquaria trade) should be aware of the issues and regulations. Enforcement presence in these trades could also be effective. Where systems already contain invasive species, an evaluation should be undertaken on a case-by-case basis to prioritize systems for management action.

7.2.3 Natural Dispersal from Other Jurisdictions

A growing number of fish species enter BC via natural northward dispersal from US waters. Species currently known to have dispersed from the US in this way include American shad, walleye, and probably yellow bullhead and brook trout. Freshwaters in the western US are now inundated with non-native fish species (e.g., Schade and Boner 2005), and reports of new species are frequent. BC cannot be isolated from these introductions, particularly in transboundary water systems such as Kootenay, Columbia, Okanagan and Skagit rivers. Thus, fisheries managers should make themselves aware of non-native species occurrences south of the border to provide a hint of the species that may be introduced in BC and their potential impacts.

7.3 Conclusions

It has become virtually impossible to ensure with complete certainty that a waterbody will remain 100% free of non-native species. Even geographic isolation can be overcome by naturally dispersing aquatic species or by humans wishing to introduce certain species. Some feel it is only a "matter of time" before BC is seriously impacted economically, environmentally and socially by non-native species. However, this argument is defeatist and works against those trying to control and prevent impacts from non-native species. Prevention and control are likely to be useful in many regions in BC. Some waterbodies, such as urban lakes and streams and large transboundary watersheds such as the Columbia River are problematic for management. In such cases impacts from non-native species are expected to grow and are almost impossible to contain. But many of BC's watersheds, particularly in the northern

regions, are fairly remote and isolated from source populations. Actions taken to stem the flow of deliberate and unintentional introductions could pay dividends in maintaining these watersheds as native species refugia. Thus we must take steps to ensure that the enforcement, trade and general public groups, as well as our governing bodies, understand the scope of this growing problem and support efforts to protect native fish values in BC.

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