THE ECOLOGICAL AND TOXICOLOGICAL SIGNIFICANCE OF ALTITUDINAL MIGRATION BY THE AMERICAN DIPPER (*CINCLUS MEXICANUS*)

by

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The ecological and toxicological significance of altitudinal migration by the American Dipper (*Cinclus mexicanus).*

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ABSTRACT

American dippers (*Cinclus mexicanus*) appear to concentrate at low elevation streams in fall and winter, implying some degree of migratory behavior in this species. Knowledge about the composition, origin and movement patterns of dipper populations are necessary to use this species as an effective biomonitor of trace contaminants in watersheds. During 1999- 2002, I color banded 522 dippers and radio-tagged 14 in the Chilliwack River watershed, British Columbia, to follow their movements during the breeding and non-breeding seasons. During the annual cycle, most individuals (84-89 %) moved from the main river to higher elevation tributaries to breed, while small numbers (11-16%) remained resident on the main river year round. Resident birds on the river initiated nests earlier, had greater proportions of second broods, and benefited from marginally higher rates of nest success and annual productivity than migrants on creeks. Later initiation of breeding by altitudinal migrants reduced productivity by increasing predation or flooding risk and prevented opportunities for initiating second clutches.

Residue analysis of dipper eggs and prey items (benthic invertebrates and salmon fry) revealed DDE, hexachlorobenzene, trans-nonachlor and PCBs as the most prevalent organic contaminants detected. PCB and organochlorine patterns were highly consistent across sites in the Georgia Basin and between dipper eggs and prey, indicating common atmospheric sources to the region that were biomagnified in lotic food chains. Organochlorines, PCBs and mercury were all significantly higher in eggs from river residents compared to creek migrants. Feathers of residents also had higher concentrations of mercury, cadmium, and copper relative to migrants. However, levels of

organic and metal contaminants in invertebrates differed minimally between the river and creeks.

Observed differences in contaminant concentrations were primarily related to the proportion of salmon fry in dipper diets. Stable isotope analysis revealed that residents consumed significantly more fish than migrants. Blood $\delta^{15}N$ values were positively correlated with total organochlorines and total DDT in eggs suggesting that fish were the primary source of organic contamination. However, daily exposure to metals that exceeded tolerable daily intake levels (selenium, aluminum and zinc) was of greater concern to migrants consuming primarily invertebrate diets.

DEDICATION

...For my mother and father, who have supported me through all my endeavors, taught me many life lessons, and the benefits of hard work.

...And for Jason, who has been an endless source of love and encouragement, my foundation for learning.

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Over the course of the 4-year study, the field component took up almost 2 full years. I wish to give special thanks to my fellow "dipper chicks" Ingrid Pollet, René McKibbin and Holly Middleton, who often risked life over limb in the field and deserve credit for much of the data presented here. Both Ingrid and René became passionate in the dipper study often taking time away from their other work to volunteer on winter surveys. I not only appreciate their hard work in the field, but their dedication to the project, and their continuing friendship. Several others volunteered with field data collection including Jason Morrissey, Harp Gill, Sandi Lee, Pippa Sheppard, Darren Fergusen, David Vockeroth, Nicole Tennant, Brent Matsuda, and Gabi Kardosi. I further wish to thank the Chilliwack River Rafting Company especially Russ Brown for "saving" us on many occasions by paddling to many difficult nests during the study. I am also grateful to the many Canadian Wildlife Service staff including Sandi Lee, Laurie Wilson, Barry Smith, Jason Komaromi, Steve Shisko, Sean Boyd, André Breault, and Wendy Easton for advice and assistance on many details of the project from banding to mounting radios, sample storage and shipments, as well as statistical and GIS help.

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I save a special place here at the end to thank my family and friends who have probably done the most to support me in all aspects of my academic endeavors. I especially thank my husband, Jason, who has done more than his share of research volunteer work, but mainly for his continuous love and support. Without them, this thesis would not have been possible...

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CHAPTER 1:

GENERAL INTRODUCTION

Rationale for the study

Lotic habitats are biologically rich and complex, supporting a variety of flora and fauna that are unique to these systems. They also supply an important freshwater resource for industrial, domestic and agricultural processes. World-wide, rivers and streams drain an area of land approximately 150 million km^2, containing a volume of water in excess of 2000 km^3 (Giller and Malmqvist 2000). Although this seems like a large amount; it represents less than 0.01 % of the world's freshwater supplies, proving lotic environments are an important habitat requiring protection.

Protection of running water habitat is not limited to the area between the stream banks since hydrological processes link streams to the soils, vegetation, rocks and atmosphere surrounding them. Human activity frequently disturbs the dynamic links between the aquatic, terrestrial and atmospheric environments within watersheds. Human influence has taken a variety of forms including eutrophication, acidification, destruction of riparian areas, changes in stream flow, as well as pollution from local and long range sources (Giller and Malmqvist 2000, Maybeck 2001). While many of these changes are temporary, others exhibit long-term effects on water chemistry and biota occupying these habitats (Maybeck 2001, Arnason and Fletcher 2003, Stewart and Skousen 2003).

Because the aquatic, terrestrial and atmospheric environments are linked within watersheds, an extremely broad range of pollutants, both organic and inorganic, can enter streams from a number of sources. This makes quantifying and measuring the effects of pollutants on numerous riparian species an extremely complex and challenging process. Many pollution abatement programs seek to employ a suitable indicator or biological

monitor that is representative of the health of the whole ecosystem. Ecological parameters such as survival, reproduction, behavior and other population characteristics can then be measured on this single species or group to assess the effects of local and long-range pollution to whole populations or communities.

It has long been recognized that birds are highly sensitive to pollution and deleterious changes to their habitat serving as early warnings of environmental degradation (Hickey 1969, Peakall et al. 1973, Mineau et al. 1984). They represent a position at the top of food chains and have relatively long life spans, thus reflecting pollutant levels in whole ecosystems (Burger 1993, Furness 1993). As a result, they have proven very useful in studying the effects of pollutants in a variety of ecosystems and their changes over time. A primary objective in ecotoxicology is to link the impacts of pollution on individual organisms to the impacts on populations and communities. However, the greatest challenge in using birds as biomonitors of pollutants is being able to disentangle interactions of pollutants and other population density stressors. This requires intimate knowledge of the ecology of the species involved.

Although many species of birds occupy river systems and may be used as suitable biomonitors of watersheds, relatively few avian species are exclusively aquatic and occupy running waters year round. Truly lotic birds are limited to the dippers (*Cinclidae*) and a few waterfowl species. Other river birds including kingfishers, sandpipers and swallows may only use the margins of streams, may be found in other habitats or only use lotic habitats seasonally. The dipper's year round residency in mountain streams and exclusively aquatic diet makes them well suited to studying the effects of pollution to watersheds as a whole.

In Europe, dippers (*Cinclus cinclus*) have been successfully used as environmental monitors of changes in water quality through acidification (Buckton et al. 1988, Ormerod et al. 1991, Vickery 1992, Logie 1995). In North America, there is also a considerable growing interest in using the American dipper (*Cinclus mexicanus)* as a biological indicator of water quality (Osborn 1999, Strom 2001, Feck 2002). However, within the Georgia Basin as with many other regions in western North America, populations of dippers appear much more mobile and are less well studied compared to the European dipper. For example, in the south coastal region of British Columbia, rivers and streams appear to host high densities of dippers in the fall and winter months with numbers seeming to far exceed resident, breeding populations (W. Campbell pers. comm.). The majority of present literature suggests that American dippers are resident (non-migratory), possibly making some seasonal altitudinal movements. However, others suggested that the large numbers of dippers that winter within this region of B.C. are from more northern latitudes subject to freezing. Lack of research in this area has eluded our understanding of the population dynamics of the American dipper in B.C. thus preventing future examination of the influence of contaminants on populations.

If the American dipper is to be successfully used as an indicator of changes in water quality through exposure to contaminants, questions related to understanding the fundamental population ecology of these birds need to be addressed. These include aspects of the population structure, seasonal distribution within the habitat, changes in abundance, patterns of movement, diet, and fecundity. Across its range, American dippers have shown an overall decline in numbers, particularly in northern regions (Sauer et al. 2003). However, lack of knowledge about dipper ecology prohibits making a

causal link between population declines and habitat destruction. For example, seasonal shifts in the distribution of populations may potentially influence local population densities, or differences in diet may contribute to large variation in the observed levels of contaminants. Thus, there is a general need to address the questions surrounding American dipper population structure and migratory patterns in order to link this knowledge to identify effects on reproduction, diet and exposure to contaminants.

Dispersal patterns in population studies

Dispersal or migration is a central concept in demographic studies concerned with regulation of populations and conservation management (Gadgil 1971, Greenwood and Harvey 1982). Several factors such as food availability, competition, predation pressure, temperature and other environmental variables are known to influence animal movements between habitats. In addition, the characteristics of a population's structure (e.g. spatial, age and sex) can also be important in determining dispersal patterns (Clobert and Lebreton 1991). These regular migratory movements have selective value in enhancing individual survival and reproductive potential between seasons (Dobson and Jones 1985, Krebs 1994).

Most studies examining bird populations generally consider natal dispersal and long-distance migration between the breeding and wintering sites as important forms of movement (Greenwood and Harvey 1982, Berthold 1996). However, other seasonal movements in and out of a study area to access better feeding areas or to avoid ice or snow cover can also strongly influence the dynamics of local populations. Altitudinal or vertical migration is a common form of periodic seasonal movement in many mountainous species that often move short distances to lowlands in winter, but routinely

migrate to higher elevations to breed (Berthold 1996). Periodic movements can cause rapid or pronounced changes in local density that can surpass those caused by fluctuations in fecundity or survival (Newton 1991). Therefore, studying movement patterns and its relationship to spatial variation in demography can have significant importance in many population studies. As such, I consider the importance of dispersal and differential movement patterns as a key aspect of American dipper population ecology, which may further influence this species' productivity, diet and exposure to contaminants.

Selection of study site

The Chilliwack watershed was selected as a study site for studying the American dipper's population structure, movement patterns, and exposure to contaminants for several key reasons. First, the Chilliwack River valley is located within the Georgia Basin, British Columbia, an area that is under pressure from rapid urban, industrial and agricultural development throughout the region. Second, Chilliwack is located in the heart of the Fraser Valley, where inorganic and organic air pollution from local and long-range sources was hypothesized as a potential source of contamination to the watershed. Third, surveys conducted by the author in 1998 in this watershed along with several other sites in the region, revealed a particularly large dipper population existed on the Chilliwack River that would support an intensive field study and was consistent with observations of having high winter densities of dippers with no knowledge about their breeding origins. Finally, the watershed was easily accessible by a paved road, which follows most of the length of the main river and a network of logging roads along several

tributaries. Therefore, good accessibility prevented several logistical constraints associated with a large-scale study of this kind.

Research Objectives

The main objectives of this research were to use a population of American dippers to:

1) identify the population's structure and seasonal migratory patterns,
2) link migratory strategy with effects on breeding performance,
3) link migratory strategy to relative composition of the diet,
4) determine sources and exposure to organic and inorganic contaminants based on new knowledge of migratory patterns and diet,
5) assess toxicological risk and potential effects associated with observed contaminant exposure.

Outline of Thesis

The thesis has been structured into five main body chapters that combine data on the American dipper's population ecology and exposure to contaminants from 4 years of field studies and laboratory analyses. In the second chapter, data is summarized from a large mark-resighting effort where over 500 dippers were individually color banded and followed in both the breeding and non-breeding seasons. The banding study was used in combination with a small-scale radio telemetry study and three years of routine river surveys to reveal the general structure of the dipper population and their seasonal migratory movements. In addition, I demonstrate how the high winter densities can be advantageously used to estimate dipper population size in this watershed and as a model for other watersheds. My overall objective was to show how different migratory

strategies could influence assessments of habitat quality using indicator species such as the dipper.

The third chapter deals with the effects of different migratory strategies (resident and altitudinal migrant) on dipper breeding performance. After following almost 100 breeding pairs over a 3-year period, I was able to assess the effects of migration, altitude, habitat and nesting substrates on overall reproductive timing and productivity. The goal was to better understand the effect of different migratory strategies on the dipper's breeding biology.

In the fourth chapter, I identify the major sources and patterns of persistent ubiquitous contaminants such as organochlorines and PCBs to dippers in the study watershed and compare these with three other sites in the Georgia Basin. Through analysis of eggs and prey items of the American dipper, I was able to determine patterns of atmospheric deposition to the watershed. In addition, biomagnification factors (BMF) were calculated to assess the ability of long range transported contaminants to accumulate in indicator species such as the dipper from its aquatic diet.

Chapter 5 presents the effects of different migratory strategies on diet composition and exposure to contaminants. A comparison of resident and migrant egg contaminant levels reveals the influence of both diet and breeding location on exposure to organochlorines, PCBs and mercury. Stable isotope techniques were further used to identify the major contributions of fish and invertebrate prey to the diets of residents and migrants and link these to observed contaminant concentrations in dipper eggs.

In Chapter 6, I quantify the degree of metal contamination to resident and migrant American dippers in the Chilliwack watershed using their diet, feathers and feces. I

present mass balance exposure models incorporating metal levels from the prey items to determine daily exposure to nine different trace metals based on differences in diet and breeding location. These models were used to interpret observed metal levels in feathers and assess toxicological risks for the Chilliwack dipper population. They also serve as an example for how to assess risk to other birds in metal contaminated systems.

The final chapter is a summary of the major findings and a synergistic interpretation of the results. I include general and specific conclusions about dipper migration and its overall influence on the ecological and toxicological parameters measured in this study. In applying my research to conservation objectives, I present a number of management recommendations for protection of watersheds and regional dipper populations. In addition, I make suggestions for future research from key questions that arose out of the present study.

Authorship

This research is a compilation of 4½ years of work primarily conducted by the author. Each main chapter is written in a manuscript style for submission to selected journals in the field. My supervisors Dr. L. Bendell-Young and Dr. J. Elliott coauthored chapters 2, 4, 5, and 6. However, the primary author (CAM) designed the study, conducted all the field research, sample collection and preparation, data analysis, and writing of the thesis. My supervisors played a role in providing research funding, advice on study design, mentoring the interpretation the results and scientific content, and reviewing drafts of the manuscripts written by the primary author. Several individuals also played important roles in assisting with data collection, laboratory analyses, advice

on study design or data analysis, and reviewing manuscript drafts. These individuals are subsequently acknowledged at the end of each chapter.

Literature Cited

Arnason, J.G. and Fletcher, B.A. 2003. A 40+ year record of Cd, Hg, Pb and U deposition in sediments of Patroon Reservoir, Albany County, NY, USA. Environmental Pollution 123: 383-391.

Berthold, P. 1996. Control of Bird Migration. Chapman and Hall, London, UK.

Buckton, S.T., Brewin, P.A., Lewis, A., Stevens, P., and Ormerod, S.J. 1998. The distribution of dippers, *Cinclus cinclus* (L.), in the acid-sensitive region of Wales, 1984-95. Freshwater Biology 39: 387-396.

Burger, J. 1993. Metals in avian feathers: bioindicators of environmental pollution. Reviews of Environmental Toxicology 5: 203-311.

Clobert, J. and Lebreton, J.D. 1991. Estimation of demographic parameters in bird populations. *In* Bird Population Studies: Relevance to conservation and management (eds. C.M. Perrins, J.D. Lebreton, G.J.M. Hirons). Oxford University Press, Oxford.

Dobson, F.S. and Jones, W.T. 1985. Multiple causes of dispersal. American Naturalist 126: 855-858.

Feck, J.M. 2002. Assessment of the American dipper (*Cinclus mexicanus*) as a biological indicator of water quality. MSc. thesis. University of Wyoming.

Furness, R.W. 1993. Birds as monitors of pollutants. *In* Birds as Monitors of Environmental Change (eds. R.W. Furness and J.J. D. Greenwood). Chapman and Hall, London, pp. 86-143.

Giller, P.S. and Malmqvist, B. 2000. The Biology of Streams and Rivers. Oxford University Press, New York.

Gadgil, M.E. 1987. Spatial structure and population vulnerability. *In* Viable populations for conservation (ed. M.E. Soulé). Cambridge University Press, Cambridge.

Greenwood, P.J. and Harvey, P.H. 1982. Natal and breeding dispersal of birds. Annual Review of Ecology and Systematics 13: 1-21.

Hickey, J.J. 1969. The Peregrine Falcon populations, their biology and decline. University of Wisconsin Press, Madison.

Krebs, C.J. 1994. Ecology: The experimental analysis of distribution and abundance- 4th edition. HarperCollins College Publishers, New York, NY.

Logie, J.W. 1995. Effects of stream acidity on non-breeding Dippers *Cinclus cinclus* in the south-central highlands of Scotland. Aquatic Conservation: Marine and Freshwater Ecosystems 5: 25-35.

Maybeck, M. 2001. Riverine quality at the Anthropocene: porpositions for a global space and time analysis, illustrated by the Seine River. Aquatic Sciences 64: 376-393.

Mineau, P., Fox, R.J., Norstrom, R.J., Weseloh, D.V., Hallett, D.J., and Ellenton, J.A. 1984. Using the Herring Gull to monitor levels and effects of organochlorine contamination in the Canadian Great Lakes. Advances in Environmental Science and Technology 14: 425-452.

Newton, I. 1991. Population limitation in birds of prey: a comparative approach. *In* Bird Population Studies: Relevance to conservation and management (eds. C.M. Perrins, J.D. Lebreton, G.J.M. Hirons). Oxford University Press, Oxford.

Ormerod, S.J., O'Halloran, J., Gribbin, S.D. and Tyler, S.J. 1991. The ecology of Dippers *Cinclus cinclus* in relation to stream acidity in upland Wales: breeding performance, calcium physiology and nestling growth. Journal of Applied Ecology 28: 419-433.

Osborn, S.A.H. 1999. Factors affecting the distribution and productivity of the American dipper (*Cinclus mexicanus*) in western Montana: does streamside development play a role? MSc. Thesis. University of Montana.

Peakall, D.B., Lincer, J.L., Risebrough, R.W., Pritchard, J.B., and Kinter, W.B. 1973. DDE- induced egg-shell thinning: structural and physiological effects in three species. Comparative Genetics and Pharmacology 4: 305-313.

Sauer, J. R., J. E. Hines, and J. Fallon. 2003. The North American Breeding Bird Survey, Results and Analysis 1966 - 2002. Version 2003.1, USGS Patuxent Wildlife Research Center, Laurel, MD.

Stewart, J. and Skousen, J. 2003. Water quality changes in a polluted stream over a 25 year period. Journal of Environmental Quality 32: 654-661.

Strom, S.M. 2000. The utility of metal biomarkers in assessing the toxicity of metals in the American dipper (*Cinclus mexicanus*). MSc. Thesis, Colorado State University.

Vickery, J. 1992. The reproductive success of the dipper *Cinclus cinclus* in relation to the acidity of streams in south-west Scotland. Freshwater Biology 28: 195-205.

CHAPTER 2:

SEASONAL TRENDS IN POPULATION DENSITY, DISTRIBUTION AND MOVEMENT OF AMERICAN DIPPERS WITHIN A WATERSHED OF SOUTHWESTERN BRITISH COLUMBIA, CANADA

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Abstract.

American Dippers concentrate on low elevation streams during fall and winter months in many parts of its range. The large influx in numbers implies some degree of migratory behavior for this species; however the breeding origin relative to the wintering location of these birds is poorly understood. Our objectives were to identify the seasonal changes in the density and distribution of American Dippers in a coastal watershed; to classify the composition of the local population; and to determine the origin and movement patterns of winter migratory birds. During 1999-2002, we color banded 522 dippers and radio-tagged 14 in the Chilliwack River watershed in southwestern British Columbia, Canada. Peak densities on the main river occurred in early November (mean = 9.8 birds/km), which was nearly 5 times higher than in early July (mean = 2.1 birds/km). The study revealed that the majority (84 % - 89 %) of the dipper population seasonally migrated, primarily moving from the low elevation river in fall and winter to the higher elevation creeks in spring. The remaining dipper population (11 % - 16 %) remained resident on the river year-round. Winter migrants showed a high degree of winter site fidelity with 67 % returning to the same site on the river for 2 or more years. Given the population of dippers in the watershed was concentrated in winter, we estimated the size of the Chilliwack watershed population from November survey data to be 429 ± 64 birds in 1999, 682 ± 79 birds in 2000, 697 ± 123 birds in 2001 and 550 ± 72 birds in 2002. This study has implications for applying American Dipper populations for use as indicators of water and habitat quality in mountainous watersheds given the population's defined structure and predictable patterns of movement between seasons.

Key words: altitudinal migration, American Dipper, Cinclus mexicanus, *indicator species, mark recapture, radio-telemetry, watershed.*

INTRODUCTION

Monitoring programs involving birds as indicators of habitat quality are now widely used on both a local and national scale as an effective way to measure spatial and temporal trends in degradation of key ecosystems (Furness 1993). In general, knowledge about the ecological impacts on study populations used in monitoring studies may serve as an early warning of habitat degradation. However, a solid understanding of the basic ecology of an indicator species is necessary to effectively interpret population level effects and monitor key ecosystems.

Numerous European studies have successfully used the dipper (*Cinclus cinclus*), an aquatic songbird, as an environmental monitor of riparian habitat, including changes in water quality (Ormerod et al. 1991, Tyler and Ormerod 1992, Logie et al. 1996, Buckton et al. 1998, Sorace et al 2002). In North America, there are also recent studies supporting the use of the American Dipper (*Cinclus mexicanus*) as a biological indicator of stream condition (Feck 2002, Strom et al. 2002). The use of an indicator or sentinel species typically assumes that it is relatively sedentary or its ecology should be well understood (Moriarty 1999). However, relatively little is known about the ecology of the North American species, compared to the well-studied Eurasian Dippers (Tyler and Ormerod 1994). In particular, American Dippers throughout their range appear much more mobile than many European populations. If the American Dipper is to be successfully used as an indicator of changes in environmental quality, questions related to understanding the population's structure and movement patterns need to be addressed.

Several studies on the Eurasian Dipper and American Dipper have alluded to the fact that *Cinclus* is a partial migrant, but most accounts have been primarily descriptive

or speculative in nature (Bent 1948, Bakus 1959a, b, Balát 1962, Jost 1969, Whitney and Whitney 1972, Tyler and Ormerod 1994). Furthermore, the extent of the migration and the location of the wintering and breeding sites of individual birds have never been examined in the North American species. In Canada, few records exist on the American Dipper's abundance and no comprehensive ecological studies have been done on this species in British Columbia, despite the fact that the majority of the Canadian population exists here (Campbell et al. 1997). However, observational data have indicated high densities of dippers occur in fall and winter months on rivers and streams of the southwestern region of the province which is consistent with reports from elsewhere in its range (King et al. 1973, Campbell et al. 1997, Kingery 1996). Speculations as to the origin of these wintering birds include a possible migration from more northern latitudes where seasonal freezing is common (Campbell et al. 1997) or an altitudinal migration from the remote mountain streams (Kingery 1996). Lack of research in this area has precluded our understanding of the population dynamics and movement patterns of the American Dipper. However, the use of marked individuals can allow us to relate wintering populations to specific breeding origins.

This research represents an important case study to address several questions about the migratory movements of dippers and the temporal trends in population density, distribution and size. Our objectives were to document the seasonal changes in density and distribution of dippers within a watershed of southwestern British Columbia; to identify the composition and size of the local population; and to determine the origin and movement patterns of winter migratory birds through extensive banding and radio telemetry methods.

METHODS

STUDY AREA

The study area was located within the Chilliwack River watershed (49°0'N, 121°4'W), a tributary of the large Fraser River system, located approximately 100 km east of Vancouver in the Cascade Mountain Range of southwestern British Columbia, Canada (Fig. 1). The Chilliwack River is 43.5 km in length fed by a large glacial lake at its upper end and merges into the Vedder River downstream. The watershed has an elevation range from 20 to 2500 m and drains an area over 1200 km^2. The watershed tributaries are first through third order streams, and the river is dominantly a fourth or fifth order stream. This watershed served as a suitable study site because it met the criteria of having high winter densities of American Dippers as observed in many watersheds throughout the region, as well as good road accessibility. The Chilliwack River and its tributaries are characterized by highly suitable dipper habitat composed of fast moving water, a variety of riffles and pools, large boulders and cliffs, and a mixture of cobble and sand substrate.

CAPTURE AND MARKING

From 1999 to 2002, we captured and marked 522 American Dippers in our study area in accordance with federal scientific, banding and animal care permits. We established eight sites along the river that were approximately 4 km apart for trapping and surveying (Fig. 1). The majority of dippers were caught from January to March wintering on the Chilliwack River at these eight sites. Additional breeding birds were caught throughout the breeding season until late July in all areas of the watershed including the tributaries. We trapped adult and newly fledged dippers primarily by

flushing them into 6 m passerine mist nets set up over moving water in narrow channels or on edges of the river and creeks. A hand-net was used to trap a few additional birds while on the nest during incubation (females) or during nest building (males). Capture and handling of breeding birds was never observed to cause nest abandonment or hatching failure. In 1999, 2000 and 2001, nestling dippers were banded when they were 12-14 days old. Nestling ages were known since we followed breeding birds and recorded laying and hatching dates. All dippers were weighed, measured and banded with a numbered USFWS aluminum band and three colored celluloid bands. Adult dippers are sexually dimorphic in size and can be sexed by wing chord measurement (Kingery 1996) or by behavior during the breeding season.

SURVEYS

Beginning in November 1999 and then every two months (except September) until November 2002, we conducted a survey of the eight river banding sites. Surveys were routinely done within the first two weeks of the month by 2 experienced observers including the author, C. Morrissey. The distance at each of the eight sites was approximately 2 km but precise measurements were determined using rangefinders (Bushnell Yardage Pro 400, Richmond Hill, Ontario) to accurately calculate bird density. Both observers walked along the riverbank searching all channels and the lower reaches of creeks where they met the river. The same search areas were used for all subsequent surveys. Each survey took three consecutive days to complete, moving upstream for each of the eight sites such that we completed three sites each on day one and two and the last two sites were completed on day three. Although it is possible that birds could have moved between sites during subsequent days of the counts, we never encountered any

double observations of banded birds between sites during any of the surveys. Surveys were not done in early September because the majority of birds would be molting and therefore not visible for counting.

We searched for all dippers and recorded all banded and unbanded birds. In addition, if we sighted a bird but were unable to identify whether it was banded or not (usually because it was flying out of range) it was recorded as unknown but included in the total count. The unique band combinations were visible with 8x binoculars or a 10 - 40x spotting scope when the birds stood on rocks, even at distances of 100 m or more. Throughout the four-year study, and particularly during the field seasons from January to July, we searched for dippers throughout the watershed and recorded any observations of banded individuals. When we sighted a dipper, every effort was made to identify whether it was banded. We kept a history for each banded bird, which included the date and location of all sightings as well as any relevant breeding information.

RADIO-TELEMETRY

In late February and early March of 2000, we radio tagged 14 adult dippers wintering along the main river channel. Six birds had been captured and color banded in 1999 and were known to move off the river in spring, while 8 were not previously marked. Transmitters with a whip antenna (Holohil Systems Ltd.) had unique frequencies and weighed approximately 1.65 g, which was approximately 2.5 % - 3.5 % of the bird's weight. They were glued to the subsurface feathers with a waterproof epoxy (Titan Corp.) approximately 1 cm anterior of the uropygial gland, allowing the radio to fall off over time, or when the birds molted. We observed birds feeding and preening normally for several days following transmitter attachment, and resighted 4 individuals

the subsequent winter without their radios. Transmitter batteries had an expected lifespan of 12 weeks. Detection from the air was excellent but transmitters had limited range of 1-2 km on the ground due to the mountainous terrain.

We tracked dippers on the river for a few days to a few weeks until we could no longer detect a signal from the radio-tagged birds on the ground. During two flights (April 17 and May 7, 2000), we flew all sections of the watershed to the snow line to try to locate the birds with radio transmitters. The second flight confirmed bird locations and detected any further movement during the 3-week period.

MOVEMENT AND POPULATION SIZE ANALYSIS

Movement patterns were analyzed using the multi-strata model in Program Mark (White and Burnham 1999). We structured the analysis to identify probability of movement between strata for the breeding and non-breeding seasons with 95 % confidence intervals. We designated the river and creeks as independent strata and report only the most parsimonious model with the lowest Akaike Information Criteria (AIC) values.

To estimate the size of the Chilliwack River dipper population, we employed two different methods. We first used a ratio estimator (Jolly 1969, Krebs 1999) to calculate population size from the bimonthly survey data. This method did not require any band resighting information but simply used population density and an estimate of the size of the wintering area (expansion factor) to calculate total population size. Program Mark was then used to estimate the population size with the Jolly-Seber model. We used Monte Carlo simulations of analytical estimates of lambda and the covariance matrix to calculate 95 % confidence intervals for population size (N). These confidence intervals

were then compared with those estimates obtained from the survey counts to confirm their validity. Our main assumption was that the parameter uncertainty was normally distributed for the simulations.

RESULTS

DIPPER DENSITIES

Over the 3-year survey period, large fluctuations in total dipper numbers occurred seasonally on the main river channel. We observed a consistent pattern of peaking numbers in November and January followed by a decline (Fig. 2). One exception was in winter of 2002, January temperatures reached highs of 17°C and heavy rains followed by rising water levels likely drove birds off the river prematurely. Two months later, in March, temperatures plummeted well below 0°C for several days with heavy snow causing many birds to return to their wintering sites on the river. This resulted in an atypical pattern for that year with respect to survey counts. Overall, winter 1999/2000 had the lowest total numbers of birds compared to later winters.

It was clear that the Chilliwack dipper population was mobile and that high densities occurred during the non-breeding season. Dipper movements produced large changes in density (number of birds/km stream) between survey counts from November 1999 to November 2002 (Fig. 3). Some birds began leaving for their breeding sites in January, but the majority of birds left the river during March causing densities to decline until July. Peak concentrations of birds along the main river occurred in early November (mean = 9.8 birds/km), which was nearly five times higher than in early July (mean = 2.1 birds/km). The survey means were consistent between years and represented high

precision, with coefficients of variation for each survey month ranging from 6.9 % to 14.7 %.

COMPOSITION OF THE WINTER POPULATION

Of the 522 individual dippers that were captured and banded during the four-year study, 250 were adults and 272 were nestlings or fledglings (Table 1). Most adults were caught during winter on the main stem of the river, but whenever possible, breeding birds throughout the watershed were also captured to facilitate future identification. Since most of the breeding birds on the river were color banded, we were able to distinguish the birds that remained resident on or near their breeding territories year-round from the migrants and dispersing juveniles (Fig. 2). A small proportion of the wintering birds (11 % - 16 %) remained resident on the river to breed. The remainder of the large winter population (84 % - 89 %) was considerably more mobile and routinely moved off the river in early spring (presumably to breed) and returned in the fall. The migratory group consisted of returning banded adults, banded juveniles and a large number of unbanded birds. The number of residents remained relatively constant year-round, except in March when there was an increase in the number of resident birds occupying territories on the river (peak breeding density = 2.41 - 2.98 birds/km). This influx of resident breeders was not sustained throughout the season as many nests failed early and those birds were usually not relocated during that season. Ultimately, we attributed the largest fluctuation in total numbers of dippers to the arrival of non-residents in the fall and winter months (Fig. 2).

The large influx of dipper numbers in fall did not result from the production of juveniles flooding the population prior to spring dispersal. Our data demonstrate that

juveniles from the watershed did not make up the majority of the winter population. After 3 years of marking 260 nestlings and 12 fledglings (in juvenile plumage) throughout the watershed from 165 nests, banded juveniles of the year only represented 1.6 % to 5.2 % of the winter survey sightings, whereas banded adults made up 18.8 % to 25.3 %. Overall, 48 of 272 nestlings (17.6%) were resighted in their first winter or later.

DIPPER MOVEMENT PATTERNS

Of individuals marked in the first 3 years of the study (1999- 2001), 102 of 280 adults (36.4 %) were resighted for 2 or more winters. Winter migrants showed a high degree of winter site fidelity with 67 % (60 of 89) returning to the river for 2 or more years. All but two birds were resighted within 1 km of their original winter capture site. The two that were not sighted at the same location were initially captured in March and likely had already started their spring migration.

All dipper resightings throughout the watershed were categorized into 8 time periods by season (breeding or non-breeding) and year (1999-2002). Movement patterns were assessed from 668 observations of 298 banded adults during those eight time periods. Both sexes had similar patterns of movement between seasons and were lumped together in the model and reported as a single probability. Analysis of movement between river and creek strata using Program Mark (White and Burnham 1999) revealed that migrant birds moved in spring (from the non-breeding to breeding season) primarily from the river to the creeks. In contrast, the movement in the fall (from breeding to non-breeding season) was primarily from the creeks to the main river (Table 2). Residents generally remained on the river between seasons with little or no movement. Surviving juveniles that were resighted post-molt moved from their natal site in the breeding season

to a wintering location on the river or a tributary. Both male and female juveniles were found wintering primarily on the main river in a similar pattern to other adult birds. However, juvenile males had a higher probability of moving onto a creek in fall than females (Table 2). All juveniles subsequently dispersed in the spring as adults.

Of the 14 birds with radio transmitters, we successfully tracked 10 birds for the duration of the battery life. Two birds were known to have lost their radios and two birds were never relocated in subsequent searches indicating radio loss, radio malfunction, death or migration outside the study area. Beginning soon after the radios were mounted in late February and early March, several dippers were recorded off the winter site for periods of several hours usually in the morning and then returned later the same day. Of the 10 birds that retained their radio transmitters and were successfully tracked to the breeding site, the majority moved from their wintering site on the river, upstream onto a nearby creek while remaining within the watershed. The distance moved was highly variable from < 1.0 km -16.5 km (mean = 5.36 km). One dipper was even located at the snowline over the Canada – USA border. The movement of most birds was accompanied by an increase in elevation (mean = +236 m, range = 0 to 750 m). Evidence from the two aerial searches indicated that the birds did not move significantly during the 3-week interval between flights. The majority of movement occurred throughout March. By mid-April, the birds had settled on their breeding sites and did not return to the wintering area until the fall.

ESTIMATING THE POPULATION SIZE

Since the majority of the dipper population concentrated at low elevations on the main river and at the mouths of creeks in winter, a ratio estimator for density (Jolly 1969,

Krebs 1999) demonstrated significant potential as an estimate for dipper population size in the Chilliwack River watershed. There was a positive linear relationship between dipper numbers in November and survey distances at each survey site, demonstrating dipper numbers were generally correlated with stream length ($r = 0.42$, $p < 0.04$). We estimated the distance of potential dipper wintering areas including the length along the river, the lower reaches of the creeks where they met the river, and all suitable side channels of the river to be approximately 60 km. Minimum estimates of the total population size (± SE) for the Chilliwack watershed were as follows: 429 ± 64 birds in November 1999, 682 ± 79 birds in November 2000, 697 ± 123 birds in November 2001, and 550 ± 72 birds in November 2002. Coefficients of variation ranged from 11.6 % to 17.6 %. Improved accuracy would require more than eight sites be surveyed.

The Jolly-Seber (J-S) model for an open population of marked individuals was used to estimate population size for comparison with the above ratio estimator technique. Simulated 95 % confidence intervals for the population size model estimates overlapped the ratio estimates in each of 1999, 2000 and 2001 for which there was sufficient data (Table 3). However, the actual J-S estimates are not directly comparable with the ratio estimates given that the J-S model estimated population size over the entire non-breeding season and the ratio estimator uses only the 3-day survey in November.

DISCUSSION

MOVEMENT PATTERNS

American Dippers in our study watershed were primarily altitudinal migrants, moving upstream in the spring and downstream in the fall. Thus, the mobility of the Chilliwack dipper population produced large changes in density during the annual cycle.

Our accounts of peak densities (7.2 – 11.6 birds/ km) occurring during the early November surveys on the river are probably the highest on record for this species. In Boulder, Colorado, the highest densities in October varied from 1.9 – 4.7 birds/km (Price and Bock 1983). Other British Columbia streams also appear to host high winter densities but lack formal counts of this species (Christmas Bird Count data: Okanagan River, Squamish River). This pattern of migratory behavior accounts for the large changes in seasonal abundance observed in many watersheds of southwestern British Columbia.

Although most of the population seasonally migrated, 11% to 16% of the winter population remained on the river to breed with only few short absences in late summer or winter. These birds, defined here as residents, clearly did not follow the same pattern of movement as the migrants. During winter, the presence of year-round residents as well as altitudinal migrants along the main river channel indicates there are two distinct groups of birds occupying the watershed. The presence of resident and migratory American and Eurasian Dippers within a single study area has only rarely been documented elsewhere, but likely exists wherever suitable stable habitat is present (Balát 1962, Price and Bock 1983).

Altitudinal migration has been documented for several alpine tolerant bird species including dark-eyed juncos (*Junco hyemalis*) (Rabenold and Rabenold 1985), spotted owls (*Strix occidentalis*) (Laymon 1989), mountain bluebirds (*Sialia currucoides*) and white-tailed ptarmigan (*Lagopus leucurus*) (Martin 2001). However, direct evidence in the literature of seasonal altitudinal migration of American Dippers is limited. A few descriptive accounts of this species regard them as year round residents that possibly

make some short-distance movements (Kingery 1996). Price and Bock (1983) gave the only detailed record of dipper movements from a large banded population in Colorado. They found similar increases in seasonal density of dippers in fall moving through their study areas on Boulder Creek and South Boulder Creek and showed the direction of movement was primarily downstream in fall and upstream in spring. However, they were unable to determine the specific location of the wintering site for most banded birds. Several other researchers have noted similar shifts in the abundance of American Dippers during the non-breeding season (Bent 1948, Jewett et al. 1953, Bakus 1959a, b, Whitney and Whitney 1972, King et al. 1973) indicating that altitudinal migration is likely common particularly in the northern part of its range.

In central and western Europe including Britain and Ireland, most populations of dippers (*Cinclus c. gularis, hibernicus* and *aquaticus*) are largely sedentary. However, regular altitudinal movements in spring and autumn are reported for some local populations breeding at high elevations across Europe (Tyler and Ormerod 1994). In Germany, Jost (1969) reported that mainly juvenile and a few adult dippers (*Cinclus cincus aquaticus*) in his study area regularly moved between watersheds by flying over ridges. The dippers of Fenno-Scandia and northwest USSR (*Cinclus cincus cinclus*) and some birds from the Ural Mountains (*Cinclus cincus uralensis*) have been found to migrate up to 1000 km between the breeding and wintering areas (Andersson and Wester 1976). Sedentary and migratory strategies in the Eurasian Dipper appear to differ on a local scale and have likely contributed to the large number of subspecies found across Europe.

It is possible that at least a portion of the wintering population in our study area could have been from more northern latitudes or at least other watersheds. However, we have no confirmed records of our banded dippers being sighted outside the Chilliwack watershed. Two of the radio-tagged birds that were not previously known to us, remained on the river and moved very little; however, two other radio-tagged birds could not be relocated soon after transmitter attachment and were excluded from the results. These two birds may have lost their tags prematurely, the radios failed, or they died. However, another possibility is they moved much greater distances, possibly breeding outside the watershed. Large-scale movements by American Dippers are not commonly reported, but indeed American Dippers have been sighted outside the breeding range. Bent (1948) recorded one dipper on the Canadian Plains some 80 km from the mountains. Another dipper sighting on the northwest shore of Lake Superior in Minnesota was estimated to be 1400 km from the nearest breeding habitat in the Black Hills of South Dakota (Green 1970, Muelhausen 1970). Price and Bock (1983) reported the longest movement of a banded nestling was 75 km (straight line distance) from its natal site. Since only 14 birds were radio-tagged in total, those four individuals may have biased our results towards movement of shorter distances. Furthermore, there is only 1 subspecies (*Cinclus mexicanus unicolour*) of the American Dipper occupying the entire Canadian and United States range, suggesting watershed populations are probably not isolated and regular migration and juvenile dispersal likely contribute to genetic mixing.

INITIATION OF MIGRATION

Speculation as to what initiates partial migration in American Dippers has been limited. Price and Bock (1983) determined that ice at higher elevations was the ultimate

cause of downstream migration in his Colorado population. Most creeks in our study area remained open year round except at the very highest elevations yet birds moved regardless of the ice conditions. Additionally, the initiation of downstream migration from the breeding site to the wintering site occurred well before poor weather and freezing would force birds to move. One possible explanation for this behavior is the anticipation for reduction in cold stress during the winter period since ambient temperatures decrease significantly with increasing elevation (Martin 2001). Variation in climate and habitat within a few vertical kilometers can be comparable to those encountered over thousands of kilometers of latitudinal travel (Rabenold and Rabenold 1985). It would therefore be advantageous to move to lower elevations in autumn to reduce energy expenditure for thermal regulation, ultimately influencing survival. Although this may be true for some high altitude breeders, many banded dippers in our study consistently wintered on the river even though their breeding site was only a few hundred meters upstream on the creeks and not significantly higher in elevation.

An alternative explanation for the dipper's altitudinal migration is the potential increase in food supply on the main river in fall and winter, which drives the downstream migration. The Chilliwack River is a large salmon-bearing stream with a productive fish hatchery. Relatively large salmon and steelhead runs occur in the fall and winter along the main stem of the Chilliwack River with escapement records of over 500 000 salmon in 1999 (British Columbia Fisheries data). We routinely observed dippers throughout the winter, feeding on salmon eggs in addition to invertebrates. Large salmon-bearing rivers throughout the south coastal region of the province have also been observed with high winter densities of American Dippers (C. Morrissey pers. observ.). Although migration

can be energetically costly, this strategy may be advantageous to maximize access to both food and nest sites in each season.

IMPLICATIONS FOR USING DIPPERS AS BIOLOGICAL INDICATORS

Differences in migratory strategies between the residents and migrants may be ecologically relevant because residents remain stable on the breeding site whereas migrants must move and establish new territories between seasons. The presence of two distinct migratory groups within a single population has implications for using dippers as indicators of habitat quality or stream condition, since exposure to environmental stressors such as aquatic contaminants may differ between migrants and residents. Migrants are potentially using a much larger area during the course of a year whereas residents inhabit a relatively small territory. In watersheds with a source of pollution affecting a single tributary or larger downstream area, differences in habitat use during the annual cycle can result in significant variation in observed contaminant levels (Morrissey et al. submitted). In addition, resident dippers that remain on their territories year-round have the advantage of being familiar with good feeding and roosting sites in winter, possibly with improved fitness consequences. Resident dippers have also been found to initiate breeding earlier than migrants, which has an effect on their seasonal productivity (Morrissey submitted). If reproduction or survival are used as measures of environmental quality, knowledge about the differences in migratory patterns are necessary for correctly assessing the effects of habitat degradation.

Similar to many wintering waterfowl species, the majority of the dipper population within the watershed appeared to concentrate during the fall and winter months. The ratio method for estimating population size should be a useful technique for

studies on dippers since it does not require banding and can effectively count a large number of birds in a short period of time. However, if abundance is used as a measure of stream quality, one must be cognoscente of the timing of censuses to report changes in density as a result of site quality rather than those caused by annual migratory movements. Furthermore, it is important to take into account whether the survey area consists of breeding or wintering habitat, as this will also strongly influence census numbers. Therefore, future studies of American Dippers should consider the potential fluctuations in density and distribution in a watershed during the annual cycle, as these may be critical for assessing exposure and effects from environmental stressors.

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LITERATURE CITED

Andersson, S., and S. Wester. 1976. Långåterfynd av nordiska strömstarar *Cinclus c. cinclus*. Vår Fågelvärld 35: 279-286 (Swedish with English summary).

Bakus, G.J. 1959a. Observations of the life history of the Dipper in Montana. Auk 76: 190-207.

Bakus, G.J. 1959b. Territoriality, movements and population density of the Dipper in Montana. Condor 61: 410-452.

Balát, F. 1962. Distribution and movements of the dipper *Cinclus c.* aquaticus on a creek and their changes during a year. Zoological Listy 11: 131-144 (English summary).

Bent, A.C. 1948. Life histories of North American nuthatches, wrens, thrashers and their allies. US National Museum Bulletin 176: 111-130.

Buckton, S.T., P.A. Brewin, A. Lewis, P. Stevens, and S.J. Ormerod. 1998. The distribution of dippers, *Cinclus cinclus* (L.), in the acid-sensitive region of Wales, 1984-95. Freshwater Biology 39: 387-396.

Campbell, R. W., N. K. Dawe, I. McTaggart-Cowan, J. M. Cooper, G.W. Kaiser, M. C. E. McNall and G. E. J. Smith. 1997. American Dipper (*Cinclus mexicanus* swainson), p. 336-343. *In* The Birds of British Columbia, Vol. 3. UBC Press, Vancouver, B.C.

Feck, J.M. 2002. Assessment of the American dipper (*Cinclus mexicanus*) as a biological indicator of water quality. MSc. thesis. University of Wyoming, Laramie.

Furness, R.W. 1993. Birds as monitors of pollutants, p. 86-143. *In* R.W. Furness and J.J. D. Greenwood [eds.], Birds as Monitors of Environmental Change. Chapman and Hall, London.

Green, J.C. 1970. Additional Dipper sightings in 1970. Loon 42:137.

Hebert, C.E. 1998. Winter severity affects migration and contaminant accumulation in Northern Great Lakes Herring Gulls. Ecological Applications 8: 669-679.

Jewett, S.G., W.P. Taylor, W.T. Shaw, and J.W. Aldrich. 1953. Birds of Washington state. University of Washington Press, Seattle, Washington.

Jolly, G.M. 1969. Sampling methods for aerial censuses of wildlife populations. East African Agricultural and Forestry Journal 34: 46-49.

Jost, O. 1969. Uber die bedeutung der wasserscheiden beim ortswechsel der Wasseramsel (*Cinclus cinclus aquaticus*). Journal of Ornithology 110: 71-78 (German with English summary).

Krebs, C.J. 1999. Ecological Methodology. Benjamin/Cummings, California.

King, D.G., A. Grass, and K. Summers 1973. Observations on the Dipper in the Skagit Valley of British Columbia. Murrelet 54: 16-19.

Kingery, H. E. 1996. American Dipper (*Cinclus mexicanus*), p.1-28. *In* A. Poole and F. Gill [eds.], The Birds of North America, no.229. The Academy of Natural Sciences, Philadelphia, and American Ornithologists' Union, Washington, D.C.

Laymon, S.A. 1989. Altitudinal migration movements of Spotted Owls in the Sierra Nevada, California. Condor 91: 837-841.

Logie, J.W., D.M. Bryant, D.L. Howell, and J.A.Vickery. 1996. Biological significance of UK critical load exceedance estimates for flowing waters: assessments of dipper *Cinclus cinclus* populations in Scotland. Journal of Applied Ecology 33: 1065-1076.

Martin, K. 2001. Wildlife in alpine and sub-alpine habitats, p. 285-310. *In* D.H. Johnson & T.A. O'Neil [Managing directors], Wildlife-habitat relationships in Oregon and Washington, Oregon State University Press, Oregon.

Moriarty, F. 1999. Ecotoxicology: the study of pollutants in ecosystems. Academic Press, San Diego, California.

Morrissey, C.A. (submitted). Effect of altitudinal migration within a watershed on the reproductive success of American dippers. Canadian Journal of Zoology.

Morrrissey, C.A., L.I. Bendell-Young, and J.E. Elliott (submitted). Linking contaminant profiles to the diet and breeding location of American dippers using stable isotope analysis. Journal of Applied Ecology.

Muelhausen, M.H. 1970. First Dipper record for Minnesota. Loon 41: 136.

Ormerod, S.J., J.O. O'Halloran, S.D. Gribbin, and S.J. Tyler 1991. The ecology of Dippers *Cinclus cinclus* in relation to stream acidity in upland Wales: breeding performance, calcium physiology and nestling growth. Journal of Applied Ecology 28: 419-433.

Osborn, S. E. 1999. Factors affecting the distribution and productivity of the American Dipper in western Montana: does streamside development play a role? M.Sc. thesis, University of Montana, Missoula.

Price, F.E., and C.E. Bock. 1983. Population ecology of the Dipper (*Cinclus mexicanus*) in the Front Range of Colorado. Studies in Avian Biology No. 7. Cooper Ornithological Society.

Rabenold, K.N., and P. Rabenold. 1985. Variation in altitudinal migration, winter segregation, and site tenacity in two subspecies of Dark-eyed Juncos in the Southern Appalachians. Auk 102: 805-819.

Sorace, A., P. Formichetti, A. Boano, P. Andreani, C. Gramegna, and L. Mancini. 2002. The presence of a river bird, the dipper, in relation to water quality and biotic indices in central Italy. Environmental Pollution 118: 89-96.

Strom, S.M., H.S. Ramsdell, and A.S. Archuleta. 2002. Aminolevulinic acid dehydratase activity in American dippers (*Cinclus mexicanus*) from a metal-impacted stream. Environmental Toxicology and Chemistry 21: 115-120.

Tyler, S.J., and S.J. Ormerod. 1992. A review of the likely causal pathways relating the reduced density of breeding dippers *Cinclus cinclus* to the acidification of upland streams. Environmental Pollution 78: 49-55.

Tyler, S.J., and S.J. Ormerod. 1994. The Dippers. Academic Press Inc., San Diego, California.

White, G.C., and K.P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46 (Supplement): 120-138.

Whitney, M., and G. Whitney. 1972. A study of the Dippers on the St. Vrain Creek, Colorado. Western Bird Bander 47: 3-9.

FIGURE LEGENDS

FIGURE 2.1. Map of the Chilliwack watershed located in southwestern British Columbia, Canada. Symbols (▲) on the watershed map indicate the eight locations for winter banding and bimonthly surveys along the main stem of the Chilliwack River. Line across watershed map shows the Canada-USA border.

FIGURE 2.2. Mean density (± SD) of American Dippers (number of birds per kilometer of river) on the Chilliwack River, B.C. over 3 years 1999 to 2002. Data were collected during routine bimonthly surveys at eight sites on the main stem of the river.

FIGURE 2.3. Summary of bimonthly surveys on the Chilliwack River, B.C. from November 1999 to November 2002. Line graph indicates survey totals including all banded, unbanded and unidentified American Dippers on the main stem of the river. Bar graph shows relative number of resident (breeding and wintering on the river) and non-resident dippers (birds that leave the river in the breeding season).

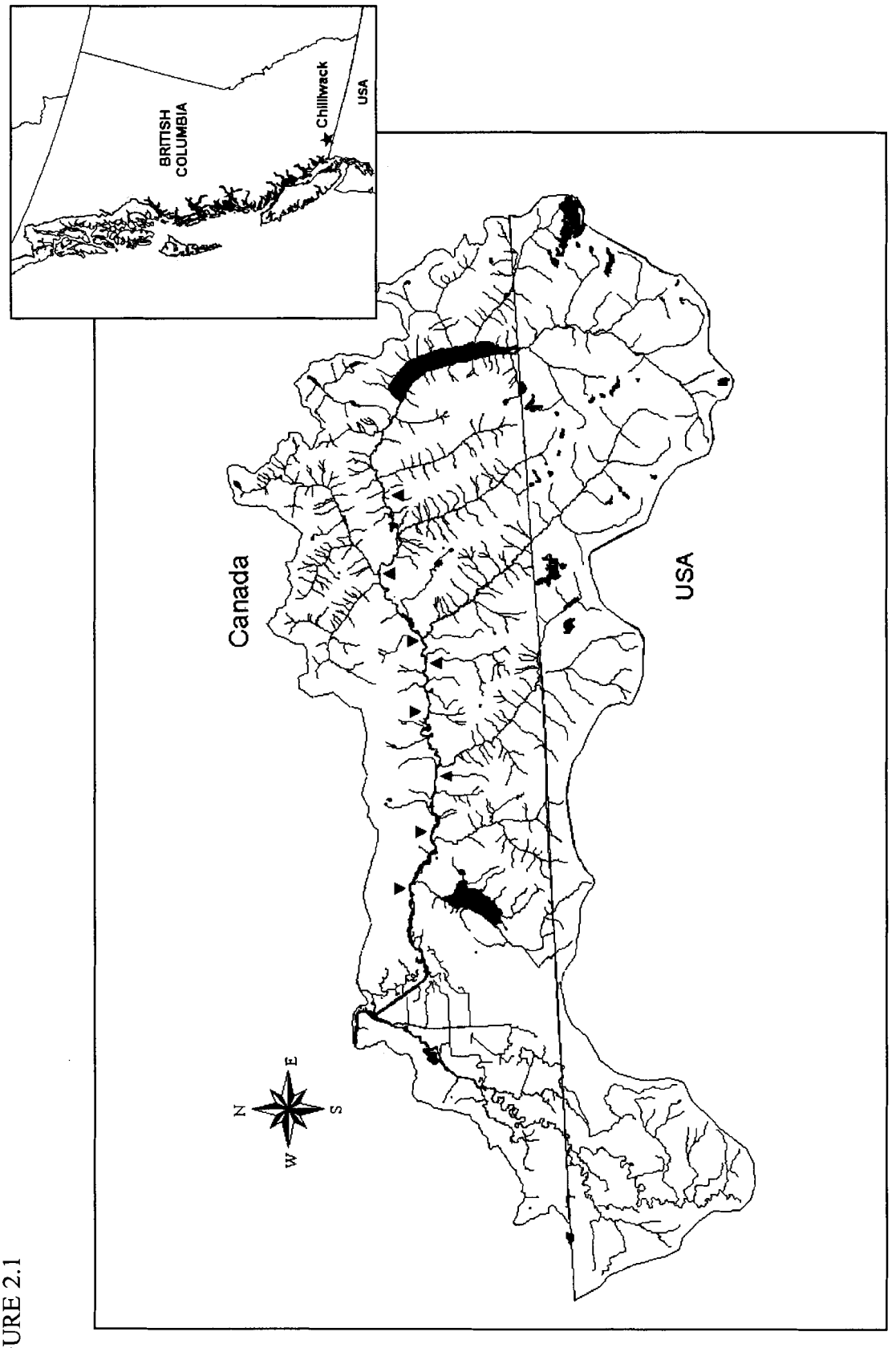

FIGURE 2.1

FIGURE 2.2

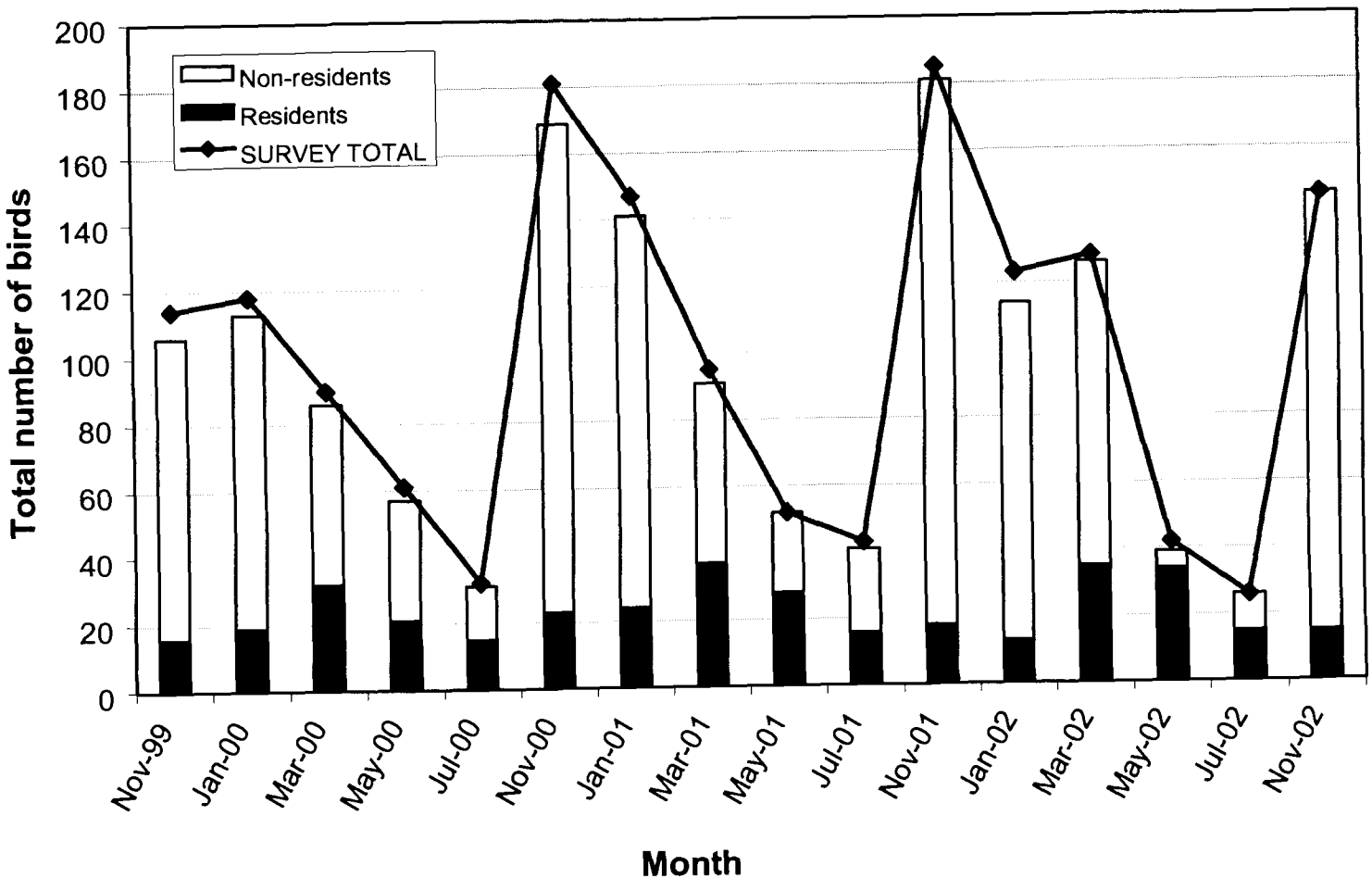
Non-residents
Residents
SURVEY TOTAL
Total number of birds
200
180
160
140
120
100
80
60
40
20
0
Nov-99
Jan-00
Mar-00
May-00
Jul-00
Nov-00
Jan-01
Mar-01
May-01
Jul-01
Nov-01
Jan-02
Mar-02
May-02
Jul-02
Nov-02
Month

FIGURE 2.3

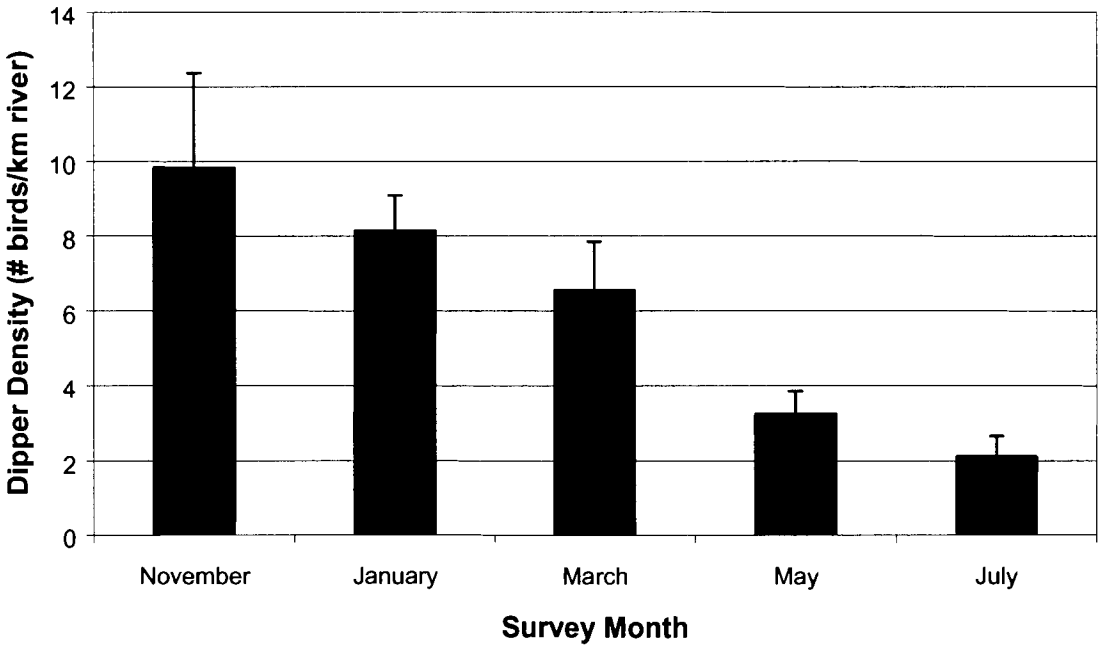
Dipper Density (# birds/km river)
14
12
10
8
6
4
2
0
November
January
March
May
July
Survey Month

TABLE 2.1. Summary of American Dipper banding effort for each age category (adult, juvenile and nestling) and year 1999-2002.

Year	Banded Adults	Banded Juveniles	Banded Nestlings	TOTAL
1999	81	7	67	155
2000	112	1	142	255
2001	39	4	51	94
2002	18	0	0	18
All Years	250	12	260	522

TABLE 2.2. Probability of movement between river and creek strata of adult and juvenile American Dippers using the multi-strata model in Program Mark (White and Burnham 1999). Data is shown for movement in the spring (from non-breeding to breeding seasons) and in the fall (from breeding to non-breeding seasons) for all years combined 1999-2002. Data on juvenile birds is not applicable (NA) in the spring since there are no juvenile birds in this period.

Movement Parameter	**Probability of Movement (95% CI)**	
	Spring (1999-2002)	**Fall (1999-2002)**
Adult (all sexes): From River to Creek	0.21 (0.09-0.43)	0.02 (0.00-0.47)
Adult (all sexes): From Creek to River	0.15 (0.01-0.75)	0.53 (0.25-0.79)
Juvenile males: From natal site to River	NA	0.60
Juvenile males: From natal site to Creek	NA	0.40
Juvenile females: From natal site to River	NA	0.88
Juvenile females: From natal site to Creek	NA	0.12

TABLE 2.3. Comparison of Chilliwack River American Dipper population size (N) estimates and 95 % confidence intervals for the ratio estimator method (Krebs 1999) and the Jolly-Seber method in Program Mark (White and Burnham 1999). Data are from surveys conducted during 3 days in November (ratio estimator) and from all mark-resighting encounters during the non-breeding period (Jolly-Seber estimator).

Survey Date	Ratio estimate N	95% CI	Season	Jolly Seber estimate N	95% CI
November 1998	No data available	No data available	Non-breeding 1998/99	321	254 - 388
November 1999	429	295 - 562	Non-breeding 1999/00	813	567 - 1059
November 2000	682	518 - 846	Non-breeding 2000/01	639	393 - 885
November 2001	697	441 – 952	Non-breeding 2001/02	425	233 - 617
November 2002	550	400 - 700	Non-breeding 2002/03	No data available	No data Available

CHAPTER 3:

EFFECT OF ALTITUDINAL MIGRATION WITHIN A WATERSHED ON THE REPRODUCTIVE SUCCESS OF AMERICAN DIPPERS

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Abstract.

Distinct changes in the distribution of American dipper (*Cinclus mexicanus*) populations within a watershed occur during the breeding season where small numbers of dippers remain resident on the main river while the majority of birds make short altitudinal movements upstream. Therefore, dippers breed over large elevation gradients within a watershed using both the main river and its associated creeks. I hypothesized that altitudinal migration would affect the timing of breeding and ultimately productivity. Additionally, since the river and its tributaries differ in habitat, elevation and nesting substrates, I hypothesized that these variables would also influence dipper breeding performance. From 1999 - 2001, I investigated American dipper reproductive success in relation to short distance migration within a watershed of British Columbia, Canada. I followed 23 dipper pairs in 1999, 40 in 2000, and 36 in 2001 of which approximately 65 % were resident pairs and 35 % were migrants. Resident pairs on the lower elevation river initiated nests earlier and had a greater proportion of second broods, contributing to slightly higher nest success and annual productivity than migrant birds on creeks or tributaries. Reduced productivity was primarily associated with later onset of breeding which increased the likelihood nests are lost to predation or flooding and reduced the probability of initiating a second clutch. Timing of breeding was affected by migratory status and year, but elevation and habitat did not directly influence breeding performance.

Introduction

Population declines in many migratory bird species has led to considerable interest in the energetic and survival costs associated with long distance migration (Klassen 1996, Sillett and Holmes 2002). Relatively few studies have examined the implications of short distance altitudinal migration in birds. This strategy may be common for many resident birds across North America, particularly alpine tolerant species (e.g. juncos, spotted owls, bluebirds, and ptarmigan) (Rabenold and Rabenold 1985, Laymon 1989, Martin 2001). Although migration is time consuming and energetically costly; this strategy may be advantageous if it allows birds to exploit areas with greater food abundance or superior nest sites in each season.

Timing of breeding can positively influence reproductive success in birds (Best and Stauffer 1980, Hartley and Shepard 1994, Kokko 1999). For some species, starting earlier increases individual productivity and offspring survival (Perrins 1965, Hochachka 1990, Verhulst and Tinbergen 1991). For birds that have multiple broods, earlier start dates may also provide an opportunity for renesting after a failed attempt or initiating a second clutch within the season (Verboven and Verhulst 1996). Several studies on passerines demonstrate that nest location within different habitats can also affect variation in breeding parameters (Klomp 1970, Krebs 1971, van Balen 1973, Martin 1988, Sanz 1998). In addition, habitat features including nest site selection are known to strongly affect breeding performance (Martin and Roper 1988, Hoover and Brittingham 1998, Regehr et al. 1998). Individuals that occupy different habitats during the breeding season are likely to have access to different resources including food and nest sites that can potentially influence their reproductive success. Therefore, the requirement of good

nesting sites and sufficient food abundance may conceivably drive some populations to migrate to higher elevations to gain access to additional breeding habitat. However, migration to those breeding sites may ultimately affect timing of breeding and overall reproductive success.

In early spring, large numbers of American dippers (*Cinclus mexicanus*) are observed moving from low elevation wintering areas on rivers to migrate upstream onto the higher elevation creeks (Price and Bock 1983, Campbell et al. 1997). This seasonal altitudinal migration is marked by a distinct change in the distribution of the population within a watershed during the breeding season where small numbers of dippers remain resident on the main river while the majority of birds make short altitudinal movements upstream (Morrissey et al. submitted). The effect of this migration is a dispersal of dippers to different elevations on both the main stem of the river and the associated creeks producing segregation between habitats.

Differences in breeding performance between resident and migrant dippers may reflect delays in breeding that are associated with the migration period or changes in habitat associated with altitude. Reproductive success is also potentially influenced by the location of nest sites within a watershed because habitat and nesting substrates differ between the river and its tributaries. Rivers are typically wider, have lower gradient with fewer cliffs and boulders than the associated creeks. In addition, the river contains habitat that is available year round without ice because of its lower elevation and higher flow. I hypothesized that the breeding performance of residents on the main river and migrants on higher elevation creeks may differ as a result of 1) migratory strategy, 2) elevation, and 3) habitat and nesting substrates. Therefore, the specific objective of this

study was to determine the influence of altitudinal migration on reproductive success of American dippers within a watershed. In this study, I used timing of breeding, clutch size, brood size, number of broods and annual female productivity as measures of breeding performance to evaluate differential reproductive success for river residents and creek migrants.

Materials and Methods

Study Area

The study was conducted in the Chilliwack River watershed (49°0'N, 121°4'W), located approximately 100 km east of Vancouver in the Cascade Mountain Range of southwestern British Columbia, Canada. The Chilliwack River watershed contains first through fourth order streams and drains an area of approximately 1200 km^2. The river is fed by a lake at its upper end and is 43.5 km in length to the Vedder Crossing where it becomes the Vedder River. The watershed has an elevation range from 20 to 2500 m with dipper nests located at 28 to 800 m. Other nests likely occurred at higher elevations but were not monitored for logistical reasons.

Data Collection

I monitored the breeding biology of American dippers within the Chilliwack watershed from early March through July 1999 to 2001 including all first, second and replacement clutches. Over the 3-year period, 99 breeding pairs including 64 resident and 35 migrant pairs were monitored at least once a week throughout the breeding season. Every attempt was made to capture and individually color band the breeding adults when the pair was identified to aid in future identification, therefore, all the pairs used in the analysis had a least one banded individual. Nests were located throughout the

watershed by searching for pairs along continuous sections of river or creeks by foot, or by following birds with nest material. All dipper nests from both the river and the creeks had an equal chance for detection using this method since these areas were searched regularly throughout the season. Not all the nests were directly accessible for the entire breeding season because of high water levels and were subsequently removed from analysis on clutch or brood size where appropriate but were monitored for timing of initiation, overall success in fledging young, and number of breeding attempts in the season. Residents and migrants were classified based on their migratory behavior and breeding location within the watershed. Migrants were generally found nesting on creeks or tributaries but wintered elsewhere either on the main river or outside the watershed, whereas residents typically nested along the main river in the same location as they were wintering (Morrissey et al. submitted). Migrant nests were monitored on seven different tributaries that were second or third order streams characterized by higher elevations with steeper gradients, fast water, a narrow path (range 3 – 20 m wide), coarse substrate and an abundance of cliffs, boulders, and debris. Resident nests were generally on a single fourth order river characterized by lower elevations, shallow gradients, slow moving reaches, a wide and channelized path (range 26 - 62 m wide), and fewer areas with cliffs or boulders. The nest sites were categorized as one of six types: cliff ledges, rock boulders, undercut stream banks with exposed tree roots, bridges or other artificial structures, fallen logs or other woody debris, and other sites (on ground under rock/concrete and one tree nest).

Dates of clutch initiation, incubation and hatching were typically known to within 1-day, or were calculated assuming one egg laid per day and a 16-day incubation period

which is consistent for this species across its range (Price and Bock 1983, Kingery 1996). Clutch size was determined after laying was complete i.e. when no new eggs were added after 2 days. Brood size was recorded during routine banding when the nestlings were 12-14 days old (halfway through the nestling stage). Nest fate was considered successful if at least one chick fledged. If no fledglings were sighted and no evidence of nest failure was apparent, the fate of the nest was recorded as unknown. I assumed the total number of fledglings from successful nests to be equal to the brood size at banding, since newly fledged dippers are difficult to locate and often leave the nest asynchronously. This assumption is supported by data on a subset of nests where detailed observations at the time of fledging show that brood size at 12-14 days typically represents the number of fledged young.

Nest failure resulted from predation, flooding, nest collapse, egg failure or nest abandonment and from unknown causes. Starvation of nestlings was never observed in this study as a cause of nest failure. I identified failure as predation when eggs or young were broken, missing, or partially eaten and the nest was damaged. Flooded nests were identified when water levels rose above the level of the nest. Nest collapse was recorded when the domed nest structure caved in but eggs were still present. Nests were generally classified as egg failure or nest abandonment if the eggs did not hatch because of infertility, insufficient incubation, or the nest was abandoned for any reason including death of an adult. If the nest was empty prior to fledging date and there was no evidence of what caused the failure, it was recorded as unknown.

Statistical Analyses

Data on clutch size, brood size, elevation and date of starting incubation were continuous scale data and normally distributed, so either an independent two-sample t-test or a one-way analysis of variance (ANOVA) with a Tukey multiple comparison test were used for comparison of means among migratory groups (river residents/ creek migrants) and years. To correct for year effects in comparing breeding start date and annual productivity for residents and migrants, we used a two-way ANOVA to include migratory status and year and report least square (LS) means. Data on nest success, multiple clutches (second and replacement), and nest site types were nominal scale data and were subsequently analyzed using chi-square contingency tables with results reported as percentages.

A stepwise regression model was used to determine which variables (year, migratory status, elevation, start date of incubating first clutches and interaction terms) were important in explaining annual productivity (number of young/ female/ season). A multiple linear regression model was then used to determine which factors affected timing of breeding. The variables that influenced predation and flooding failures were analyzed using a logistic regression model with sequential elimination of non-significant terms. The initial predation and flooding models included start date of breeding, migratory status, elevation, year and type of nest site but were reduced using stepwise procedures. Whole model results are presented where applicable in addition to the effect tests. Where comparisons were being made among migratory groups or for predictors in the regression models, data on a single pair were used only once each year by focusing on the female's first nest attempt. Data from each year were treated as independent as the

majority of pairs were only sampled in one year. Measures are presented as means ± SE unless otherwise stated. The significance level was set at 0.05 for all analyses. Statistical analyses were performed using the software JMP IN® 4.0 (SAS Institute Inc.).

Results

General Breeding Biology

The mean Julian date (1 January = day 1) for starting incubation of first clutches was 95.6 ± 1.5 days (5 April) but was significantly different among study years ($F_{2,93}$ = 7.52, P < 0.001). The harsher winter of 1999 that had record snowfall that lasted well into March, produced a later start date of incubation (104.0 ± 2.7 days = 14 April) compared to the relatively warm and dry winters of 2000 and 2001 that had earlier initiation (2000: 90.1 ± 2.3 days = 31 March, 2001: 94.6 ± 2.3 days = 5 April).

There was no effect of year on mean clutch size ($F_{2,111}$ = 1.99, P = 0.14) and mean brood size ($F_{2,125}$ = 1.89, P = 0.16) (Table 1). The proportion of second clutches and replacement clutches was also not different among years (χ^2_4 = 1.9, P = 0.8). Clutch size and brood size were not significantly different for second and replacement clutches compared to first clutches although repeat clutches tended to be smaller ($F_{2,111}$ = 1.12, P = 0.3; $F_{2,125}$ = 1.55, P = 0.2).

A total of 158 nest attempts including first, second and replacement clutches were followed over the three-year period. For all years combined, 99 nests (63 %) were first clutches, 38 (24 %) were second clutches, and 21 (13 %) were replacement of failed clutches. Overall, 59 % (n = 93) of the nesting attempts successfully fledged young, 37 % (n = 59) failed and 4 % (n = 6) were unknown (Table 1). There were no differences

among years in overall nest success ($\chi^2_4 = 3.7$, P = 0.5). Over the course of the breeding season, 73.2 % of pairs successfully fledged at least one young. All pairs combined produced an average of 3.4 ± 0.3 young per season (range 0-10) which was not significantly different among study years ($F_{2,94} = 2.18$, P = 0.12) (Table 1).

Effect of Migratory Behavior

Data for comparison of breeding performance of resident and migrant pairs are summarized in Table 2. Timing of starting incubation of first clutches for creek migrants was significantly later than residents on the river for every year of the study ($F_{3,88} = 8.76$, P< 0.0001) (Fig.1). Clutch size was not significantly different for creek migrants compared to river residents ($t_{74} = -0.93$, P = 0.4). The number of nestlings (brood size) was also not significantly different for migrants compared to residents ($t_{83} = -1.6$, P = 0.1) (Table 2). In addition, nestling mass did not differ by location when corrected for age at weighing and for the size of the brood (LS mean river: 46.9 ± 0.5 g, LS mean creek: 46.9 ± 0.7 g) ($F_1 = 0.008$, P = 0.9) (Table 2).

The individual nest success of migrant first clutches (53.1 %) appeared lower than the residents (67.2 %), although not statistically significant ($\chi^2_1 = 1.8$, P = 0.18). Residents and migrants had the same proportion of first clutches (residents 62.5 %, migrants 62.9 %); however, the proportion of second and replacement clutches between groups was different ($\chi^2_2 = 17.56$, P < 0.0002). Resident pairs had a higher proportion of second broods following successful attempts (49.2%) compared to migrants (17.6%). In addition, migrants had a greater proportion of replacement clutches following failed attempts (41.2%) compared to the residents (10.8%) (Table 2). On average, resident pairs produced 3.7 ± 0.4 young per season compared to migrants that produced 2.8 ± 0.5

young per season, after correcting for any year effects this difference was marginally insignificant ($F_{3,93} = 2.25$, $P = 0.09$) (Table 2).

A stepwise regression was used to determine which variables were important in explaining the annual productivity or number of young produced per female during the breeding season. The start date of breeding had a strong negative effect on annual productivity ($F_{3,87} = 19.5$, $P < 0.0001$) (Fig.2). Annual productivity also varied by the year of the study ($F_{3,87} = 4.3$, $P = 0.04$). Migratory status, elevation and other interaction terms had no effect. Further regression analysis of the factors that affected the start date of breeding demonstrated that both migratory status ($F = 36.2$, $P < 0.0001$) and year ($F = 6.56$, $P = 0.002$) were significant predictors of when dippers started breeding. Migrants initiated breeding later than residents (migrant LS mean = 106 ± 3 days, resident LS mean = 92 ± 2 days) and in 1999, breeding was initiated later than 2000 or 2001. Breeding location and elevation had no additional effect in determining annual productivity. Therefore, dippers that start incubating first clutches later either because of migration to the breeding sites or because of year effects resulted in lower annual productivity.

Nest Site Characteristics and Causes of Failure

The relative use of nest site types differed significantly between the migrants and residents ($\chi^2_5 = 49.3$, $P < 0.0001$) (Table 3). Specifically, migrant dippers on creeks used predominantly cliff sites (52 %) followed by logs/woody debris (22 %) and boulders (19 %) and rarely used artificial sites (2 %) or stream banks (6 %). Resident dippers on the main river used a wider variety of nest sites including artificial structures (30 %), cliffs

(26 %), stream banks (24 %), and boulders (13 %) but with no clear prevalence of use at any particular site.

The most common causes of nest failure were predation (40.7 %) and flooding (23.7 %). The cause of failure for all nest sites was not significantly different between residents and migrants ($\chi^2_4 = 6.5$, $P = 0.17$); however, flooding was notably higher for the creek migrants (33%) over the river residents (17 %). There was a general trend towards increasing risk to both predation and flooding of first breeding attempts with later breeding start dates. Early nesting birds (prior to day 88-89) avoided both predation and flooding. However, after that date, proportions of failures from these two sources increased (Fig. 3). A logistic regression model was used to determine the variables that predicted predation and flooding of first clutches. Both whole models were significant (predation model: $\chi^2_8 = 14.66$, $P = 0.04$, flooding model: $\chi^2_8 = 17.6$, $P = 0.02$). For the predation model, only start of breeding had a significant effect ($P = 0.01$), while nest site type and year also contributed to the overall model. Flooding was best predicted by start date of breeding, year, elevation and nest site, however only the start date of breeding was again a significant individual effect ($P = 0.05$). Nest site type greatly contributed to the overall strength of the flooding model. In fact, the sites that most commonly flooded were boulders (43 %), cliffs (43 %) and logs/woody debris (14 %), which corresponds to the sites primarily occupied by the later breeding creek migrants (Table 3).

Discussion

American dippers in this study area appear to have two migratory strategies: altitudinal migration to higher elevation tributaries or year round residency on the low elevation river. No previous studies of American dippers have identified differential

reproductive success at different locations within a watershed. In this study, I demonstrate that altitudinal migrants breeding on tributaries initiated breeding later, had a higher frequency of replacement clutches and fewer second broods compared to resident pairs on the main river. Timing of initiation of breeding was the most significant effect on annual productivity. However, I found no additional effect of resident or migrant status over the timing of breeding in predicting productivity. Instead, migratory status was an important predictor for timing of initiation of breeding. Since migrant nests had the same number of eggs and nestlings as those of residents, and nestling mass was identical for both locations, observed differences in productivity were not likely caused by variation in food supply. Instead, the differences are more likely caused by extrinsic factors such as flooding and predation, which resulted in a larger number of total nest failures.

Nest success of first clutches did not differ significantly between the residents and migrants, but evidence of higher nest failures for the creek migrants was observed given the larger proportion of replacement broods initiated by migrants. Resident dippers that start breeding earlier may be able to produce more young per season by initiating more second broods and having fewer replacement broods. Limitations of sample size in the migrant group (n = 35 pairs) may have decreased statistical power for the comparisons of brood size, annual productivity and success of nest attempts between residents and migrants. However, residents produced on average 3.7 nestlings versus migrants, which only produced 2.8 nestlings per year. This difference is likely biologically relevant but not detected using the present sample. Timing of breeding, however, was clearly different between the migratory groups and had a strong influence on productivity. In

fact, dippers breeding even slightly later in the season were at a greater risk for nest failures from flooding or predation. Flooding was the second most common cause of nest failure, and was found to be almost two times higher for the later breeding migrants. Furthermore, it is now widely recognized in many breeding passerines that the earliest broods generally have higher survival and higher recruitment rate (Perrins 1965, Hochachka 1990). Ormerod and Tyler (1993) determined that post fledging survival of European Dippers varied significantly during the breeding season with most survivors coming from attempts in the peak period of hatching and declined later in the season. Therefore, dippers that can begin nesting early will have the opportunity to avoid predation and flooding, produce more double broods and may also increase their individual brood's chance for survival.

Other dipper researchers have found positive relationships between elevation and timing of breeding (Bakus 1959a, Goodge 1959, Sullivan 1973, Ealey 1977, Tyler and Ormerod 1994). Nesting in dippers likely begins soon after the females have sufficient food for egg production (Perrins 1970, Drent and Daan 1980). Therefore, it is reasonable that high elevation territories will have delayed access to food resources due to ice cover and colder temperatures. However, my study found no direct effect of elevation on productivity or timing of breeding after controlling for migratory strategy. In fact, river residents occupied territories in a wide range of elevations, which overlapped those of altitudinal migrants. Therefore, delays in breeding were a result of the migration period from lower elevation wintering sites on the river to higher elevation tributaries. Elevation was not a cause of delayed breeding per se, but is a consequence of altitudinal migration. Many dippers that migrate onto creeks during the breeding season will establish

territories and find mates later than birds breeding and wintering in the same location on the river, which may be at a cost to their reproductive success.

Resident and migrant dippers that occupy river or creek habitats during the breeding season may have access to different resources including food and nest sites that can potentially influence their reproductive success. However, migration during the breeding season did not appear to be related to food supply or improved habitat quality. Migrants and residents had no difference in clutch or brood size and although food abundance was not measured directly, nestling mass was identical for both locations. Instead, it has been suggested that competition for limited nest sites may force the majority of the population to disperse over a wider area during the breeding season (Price and Bock 1983, Tyler and Ormerod 1994). Densities of breeding pairs along the Chilliwack River were relatively high (approx. 2.41 - 2.98 birds km^{-1}) (Morrissey unpubl. data) compared to reports in the literature of <1.0 - 1.7 birds km^{-1} (Price and Bock 1983) and 1.16 - 1.22 birds km^{-1} (Ealey 1977). Given the relatively high densities, many dippers may seasonally migrate onto the tributaries where nest sites such as cliffs and boulders are more abundant. This ultimately results in a segregation of birds between river and creek habitats but without any significant gain in habitat quality. Experimental increases in the number of nest sites available along the main river through addition of nest boxes would reveal whether nest site limitation is a significant cause of seasonal altitudinal migration.

Migratory status not only influenced timing of breeding, but also the distribution of nest sites used by residents and migrants. River and creek locations differ in the type of substrates available for nesting sites probably because of the differences in habitat.

The creeks had more natural cliffs and boulders and tended to be narrower with steeper gradients. Consistent with the habitat, migrant birds did use cliffs, boulders and fallen logs or woody debris with greater frequency than the resident birds. Those specific sites were also found to be more susceptible to flooding during high water events. In general, nest site type was an important variable in predicting flooding and predation events. Most other studies of dippers report a similar distribution of nest sites and common causes of nest failure, mainly due to predation and flooding (Ealey 1977, Price and Bock 1983, Osborn 1999). For many species of passerines, nest predation remains the principal source of nest losses across a broad diversity of habitats and locations, which accounts for an average of 80% of nest losses (Ricklefs 1969, Martin 1993). But for birds nesting near water, loss due to flooding is also a significant cause of nest failure (Burger 1985). Therefore, quantifying nest site availability for resident and migrant dippers occupying different habitats may be important for predicting nest losses due to flooding and predation events.

Migrant dippers nesting on creeks used artificial sites at a very low frequency (2 %) compared to river residents (30 %) despite their presence in creek habitats. Most artificial sites including bridges had a moderate success rate and were free from flooding but typically had higher predation and nest abandonment. Ealey (1977) noted no use of artificial structures despite their presence in his study area. In other studies including those on the Eurasian Dipper (*Cinclus cinclus*), bridges and other artificial sites were occupied at a higher frequency but did not show differential reproductive success (Tyler and Ormerod 1994, Smiddy et al. 1995, Osborn 1999). Shaw (1978) and Osborn (1999) found that nest success of dippers was similar at natural and artificial sites. Artificial

sites have the benefit of being protected from flood events but at a trade-off with increased predation and disturbance. It is probable that this species uses artificial sites opportunistically only when other natural nest sites are unavailable.

Overall nesting success, clutch size, brood size, and seasonal fecundity were very similar to values reported for American dippers elsewhere in the species range. Nest success of dippers in this study (58.9%) is high for passerines, but is typical for this species, which uses domed-shape nests that are well camouflaged and often inaccessible to predators. Overall breeding success for dippers in other studies, defined as percentage of nests producing at least one fledgling, ranges from 57.1 % to 80.0 % (Bakus 1959a, Sullivan 1973, Ealey 1977, Price and Bock 1983, Osborn 1999). These studies do not consider effects of migration or habitat. Therefore, future studies that assess breeding success in American dippers would benefit from accounting for potential differences due to migratory strategy.

In conclusion, altitudinal migration had a negative effect on reproductive timing and overall productivity of American dippers. Resident birds were able to gain access to mates and breeding sites significantly earlier than those that migrate onto the creeks. Thus, resident dippers had greater opportunity for avoiding predators and flood events and initiating second clutches. Although reproductive success was not directly related to migratory strategy, residents may benefit from improved survival, higher lifetime reproductive output or greater survival of their offspring. Future research on survival costs of being a permanent resident or altitudinal migrant and investigations into the factors that influence individual migration strategies, may shed greater light on our current understanding of altitudinal migration.

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References

Bakus, G.J. 1959a. Observations on the life history of the Dipper in Montana. Auk, **76**: 190-207.

Best, L.B., and Stauffer, D.F. 1980. Factors affecting nesting success in riparian bird communities. Condor, **82**: 149-158.

Burger, J. 1985. Habitat selection in temperate marsh-nesting birds. *In* Habitat selection in birds. *Edited by* M.L. Cody. Academic Press, San Diego, CA. pp. 253-278.

Campbell, R.W., Dawe, N.K., McTaggart-Cowan, I., Cooper, J.M., Kaiser, G.W., McNall, M.C.E., and Smith, G.E.J. 1997. American Dipper (*Cinclus mexicanus*

swainson). *In* The Birds of British Columbia, Vol. 3. UBC Press, Vancouver, BC. pp. 336-343.

Drent, E.H., and Daan, S. 1980. The prudent parent: Energetic adjustments in avian breeding. Ardea, **68**: 225-252.

Ealey, D.M. 1977. Aspects of the ecology and behaviour of a breeding population of dippers (*Cinclus mexicanus*: Passeriformes) in southern Alberta. M.Sc. Thesis, University of Alberta, Edmonton.

Goodge, W.R. 1959. Locomotion and other behavior of the Dipper. Condor, **61**: 4-17.

Hann, H.W. 1950. Nesting behavior of the American Dipper in Colorado. Condor, **52**: 49-62.

Hartley, I.R., and Shepard, M. 1994. Nesting success in relation to timing of breeding in the Corn Bunting on North Uist. Ardea, **82**: 173-184.

Hochachka, W. 1990. Seasonal decline in reproductive performance of song sparrows. Ecology, **71**: 1279-1288.

Hoover, J.P., and Brittingham, M.C. 1998. Nest-site selection and nesting success of Wood Thrushes. Wilson Bull. **110**: 375-383.

Kingery, H.E. 1996. American Dipper (*Cinclus mexicanus*). *In* The birds of North America, no.229. *Edited by* A.Poole and F.Gill. Academy of Natural Sciences, Philadelphia, and American Ornithologists' Union, Washington, D.C.

Klassen, M. 1996. Metabolic constraints on long-distance migration in birds. J. Experiment. Biol. **199**: 57-64.

Klomp, H. 1970. The determination of clutch size in birds. A review. Ardea, **58**: 1-124.

Kokko, H. 1999. Competition for early arrival in migratory birds. J. Anim. Ecol. **68**: 940-950.

Krebs, J.R. 1971. Territory and breeding density in the Great Tit, *Parus major*. Ecology, **52**: 2-22.

Laymon, S.A. 1989. Altitudinal migration movements of Spotted Owls in the Sierra Nevada, California. Condor, **91**: 837-841.

Martin, K. 2001. Wildlife in alpine and sub-alpine habitats. *In* Wildlife-habitat relationships in Oregon and Washington. *Managing directors* D.H. Johnson & T.A. O'Neil. Oregon State University Press, Oregon. pp 285-310.

Martin, T.E. 1988. Habitat and area effects on forest bird assemblages: Is predation an influence? Ecology, **69**: 74-84.

Martin, T.E. 1993. Nest predation and nest sites: new perspectives on old patterns. Biosci. **43**: 523-532.

Martin, T.E., and Roper, J.J. 1988. Nest predation and nest-site selection of a western population of the Hermit Thrush. Condor, **90**: 51-57.

Morrissey, C.A., L.I. Bendell-Young, and J.E. Elliott. Seasonal trends in population density, distribution and movement of American Dippers within a watershed of southwestern British Columbia, Canada. Submitted to Condor.

Ormerod, S.J., and Tyler, S.J. 1993. The adaptive significance of brood size and time of breeding in the dipper *Cinclus cinclus* (Aves: Passeriformes) as seen from post-fledging survival. J. Zool. **231**: 371-381.

Osborn, S.E. 1999. Factors affecting the distribution and productivity of the American Dipper in western Montana: does streamside development play a role? M.Sc. thesis, University of Montana, Missoula.

Perrins, C.M. 1965. Population fluctuations and clutch size in the Great tit, *Parus major L.* J. Anim. Ecol. **61**: 13-19.

Perrins, C.M. 1970. The timing of bird's breeding season. Ibis, **112**: 242-255.

Price, F.E., and Bock, C.E. 1983. Population ecology of the Dipper (*Cinclus mexicanus*) in the Front Range of Colorado. Studies in Avian Biol. No. 7. Cooper Ornithological Society.

Rabenold, K.N., and Rabenold, P. 1985. Variation in altitudinal migration, winter segregation, and site tenacity in two subspecies of Dark-eyed Juncos in the Southern Appalachians. Auk, **102**: 805-819.

Regehr, H.M., Rodway, M.S., and Montevecchi, W.A. 1998. Antipredator benefits of nest-site selection in Black-legged Kittiwakes. Can. J. Zool. **76**: 910-915.

Ricklefs, R.E. 1969. An analysis of nesting mortality in birds. Smithson. Contr. Zool. **9**: 1-48.

Sanz, J.J. 1998. Effects of geographic location and habitat on breeding parameters of Great Tits. Auk, **115**: 1034-1051.

Shaw, G. 1978. The breeding biology of the Dipper. Bird Study, **25**: 149-160.

Sillett, T.S., and Holmes, R.T. 2002. Variation in survivorship of a migratory songbird throughout its annual cycle. J. Anim. Ecol. **71**: 296-308.

Smiddy, P., O'Halloran, J., O'Mahony, B., and Taylor, A.J. 1995. The breeding biology of the Dipper *Cinclus cinclus* in south-west Ireland. Bird Study, **42**: 76-81.

Sullivan, J.O. 1973. Ecology and behavior of the Dipper, adaptations of a passerine to an aquatic environment. Ph.D. dissertation, University of Montana, Missoula.

Svensson, E., and Nilsson, J.Å. 1995. Food supply, territory quality, and reproductive timing in the Blue Tit (*Parus Caeruleus*). Ecology, **76**: 1804-1812.

Tyler, S.J., and Ormerod, S.J. 1994. The dippers. Academic Press, San Diego, CA.

van Balen, J.H. 1973. A comparative study of the breeding ecology of the Great Tit *Parus major* in different habitats. Ardea, **61**: 1-93.

Verboven, N., and Verhulst, S. 1996. Seasonal variation in the incidence of double broods: the date hypothesis fits better than the quality hypothesis. J. Anim. Ecol. **65**: 264-273.

Verhulst, S., and Tinbergen, J.M. 1991. Experimental evidence for a causal relationship between timing and success of reproduction in the great tit *Parus m. major*. J. Anim. Ecol. **60**: 269-282.

Figure Legends

Figure 3.1. Comparison of mean Julian start date (1 January = day 1) of incubation for first clutches (± SE) of residents and migrants in the Chilliwack watershed 1999, 2000 and 2001. Residents and migrants were significantly different in breeding start dates for every year ($P < 0.001$).

Figure 3.2. Annual productivity of American dippers in the Chilliwack watershed 1999- 2001 as measured by number of young produced per female during a breeding season in relation to the Julian start date of incubation for first clutches.

Figure 3.3. Percentage of American dipper nests that failed due to predation (n = 16) or flooding (n = 9) for each Julian day interval of breeding start dates. Start dates of breeding are in 10-day intervals from day 70 (earliest start date recorded) to day 137 (latest start date recorded) for initiating incubation of first clutches. Numbers above bars indicate number of nests failed in each time interval by predation or flooding.

Figure 3.1

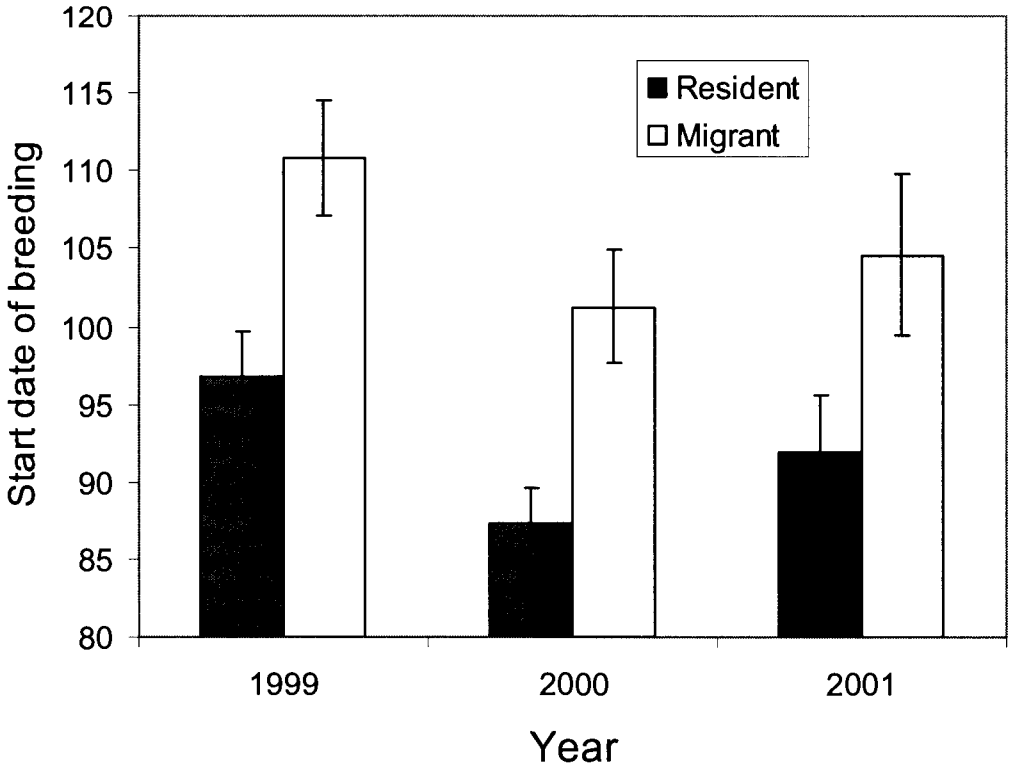
120
115
110
105
100
95
90
85
80
Start date of breeding
Resident
Migrant
1999
2000
2001
Year

Figure 3.2

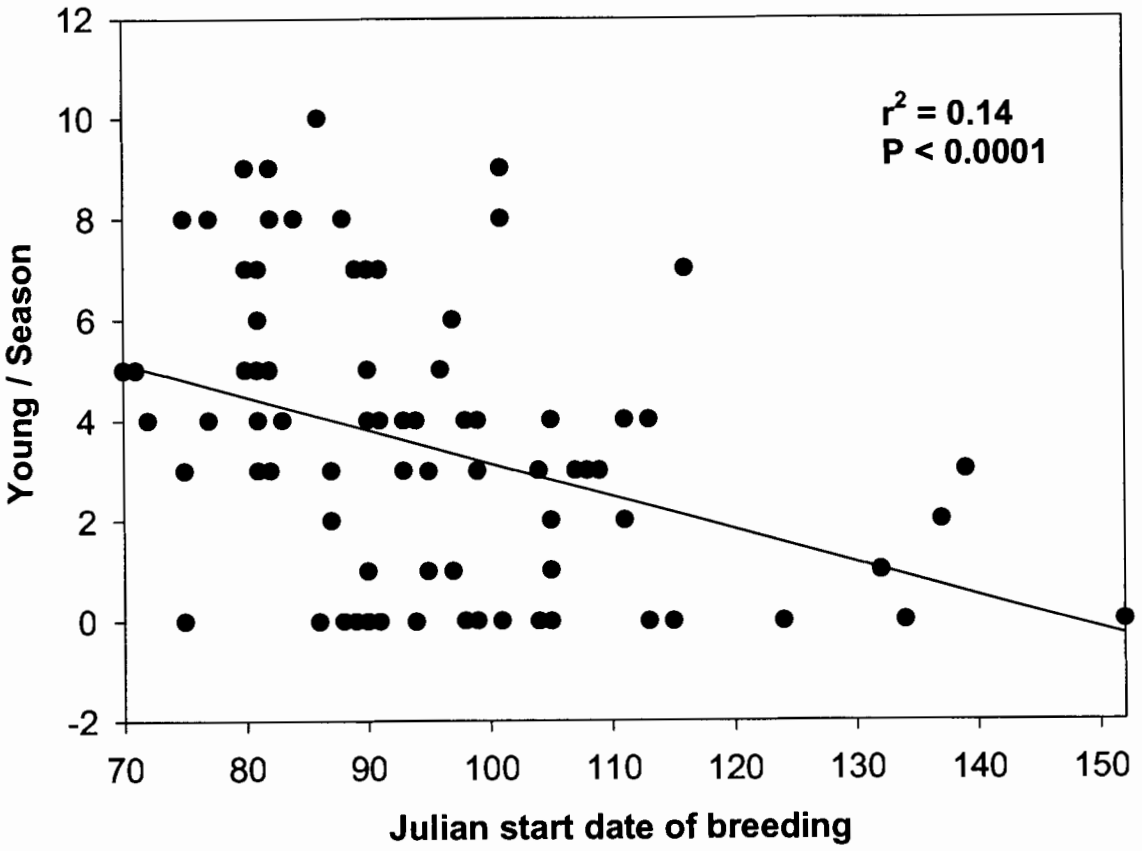
12
10
8
6
4
2
0
-2
Young / Season
$r^2 = 0.14$
$P < 0.0001$
70
80
90
100
110
120
130
140
150
Julian start date of breeding

Figure 3.3

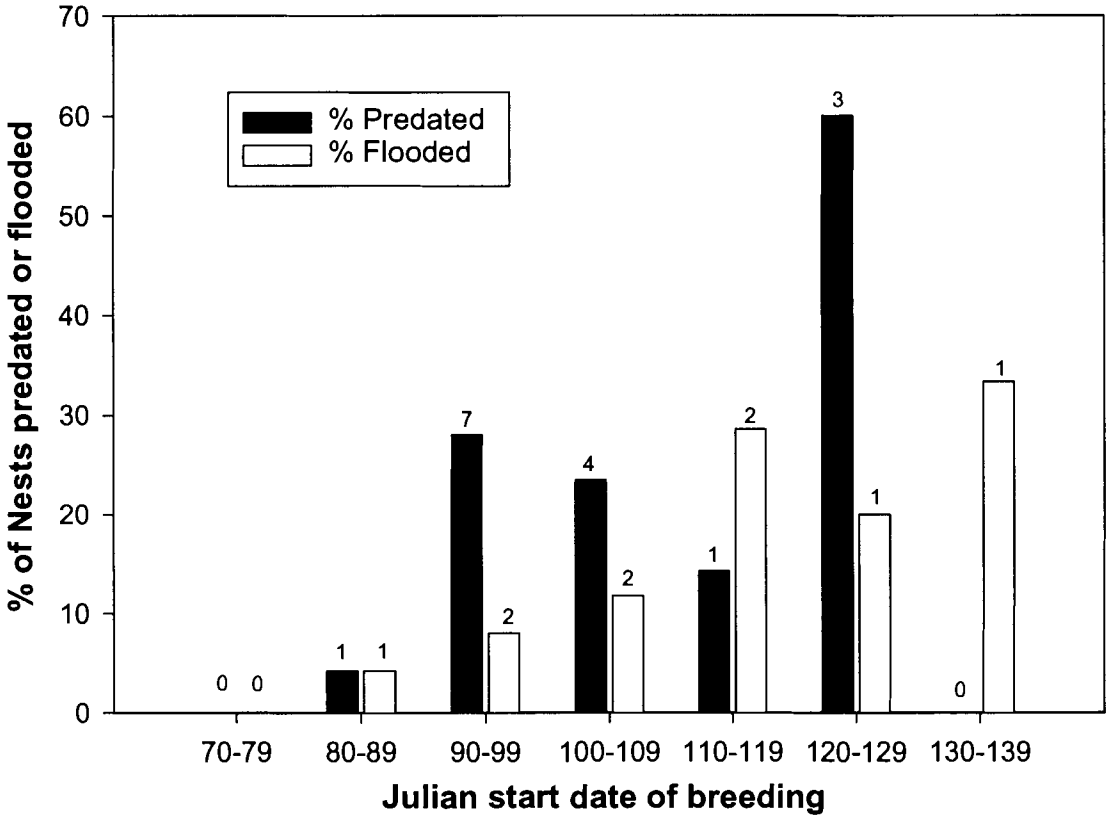
% Predated
% Flooded
% of Nests predated or flooded
Julian start date of breeding
70-79
80-89
90-99
100-109
110-119
120-129
130-139
0
10
20
30
40
50
60
70
0 0
1 1
7
2
4
2
1
2
3
1
0
1

TABLE 3.1. Summary of reproductive performance of American dippers in the Chilliwack watershed, British Columbia, Canada, 1999-2001. Data shows comparison of mean clutch and brood sizes, overall nest success for all nest attempts, and annual productivity of pairs (number of young/season). All values are given as means ± SE or percentages (n).

Year	Clutch size	Brood size	% Successful	% Failed	% Unknown	Annual Productivity
1999	4.11 ± 0.15 (28)	2.42 ± 0.34 (31)	64.9 (24)	35.1 (13)	0.0 (0)	3.39 ± 0.57 (23)
2000	4.26 ± 0.11 (50)	2.53 ± 0.24 (58)	58.5 (38)	36.9 (24)	4.6 (3)	4.00 ± 0.45 (40)
2001	4.47 ± 0.13 (36)	1.79 ± 0.30 (39)	55.4 (31)	39.3 (22)	5.4 (3)	2.67 ± 0.46 (36)
All years	4.30 ± 0.08 (114)	2.28 ± 1.9 (128)	58.9 (93)	37.3 (59)	3.8 (6)	3.36 ± 0.28 (99)

TABLE 3.2. Summary of reproductive parameters measured in comparing success of American dipper pairs of residents and migrants on the Chilliwack River and its tributaries in British Columbia, Canada, 1999-2001. Least square (LS) means reported to correct for year effects in productivity and start date of incubation and for brood size and age effects in nestling mass.

Parameters measured	Residents	Migrants	Significance (p)
N (pairs)	64	35	
Mean elevation (m)	179 ± 13	285 ± 18	<0.0001
Clutch size	4.4 ± 0.1	4.2 ± 0.1	ns
Brood size	2.7 ± 0.3	2.0 ± 0.4	ns
Nestling mass (LS mean)	46.9 ± 0.5	46.9 ± 0.7	ns
Annual productivity (LS mean)	3.7 ± 0.4	2.8 ± 0.5	ns
Start date of incubation (LS mean)	92 ± 2 (2 April)	106 ± 3 (16 April)	<0.0001
% Successful	67.2	53.1	ns
% Pairs with second clutches	49.2	17.6	< 0.0002
% Pairs with replacement clutches	10.8	41.2	< 0.0002

TABLE 3.3. Comparison of nest site success (%), relative frequency of use (%) by resident and migrant American dippers, and the proportion of failures from predation and flooding for each nest site type (%) in the Chilliwack watershed, British Columbia, Canada, 1999-2001 (see methods for explanation of nest site classification).

Nest Site Type	Nest Success (%)	Frequency of Use (%)		Cause of Nest Failure (%)	
		Residents	Migrants	Predation	Flooding
Cliffs	63.6	26	52	38	43
Stream banks/roots	67.9	24	6	25	0
Boulders	39.1	13	19	4	43
Artificial	56.3	30	2	21	0
Logs/wood debris	56.3	4	22	8	14
Other	75.0	4	0	4	0
TOTAL (All nest sites)	58.9			40.7	23.7

CHAPTER 4:

CONTRIBUTIONS FROM ATMOSPHERIC DEPOSITION TO FOOD CHAIN BIOMAGNIFICATION IN THE CHILLIWACK WATERSHED OF BRITISH COLUMBIA, CANADA

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Abstract

Research on food chain contamination in remote arctic and lake ecosystems has revealed significant concentrations of persistent organic pollutants in fish and wildlife from long-range atmospheric sources. Biota occupying mountain streams are similarly exposed to organic contaminants from atmospheric deposition through runoff of contaminated snow pack and glaciers. Therefore, we assessed the extent to which atmospheric pollutants were contributing to food chain contamination in mountain streams of southwestern British Columbia, Canada, particularly the Chilliwack River watershed. Through analysis of American dipper (*Cinclus mexicanus*) eggs, benthic invertebrates and salmon fry prey; we found that polychlorinated biphenyls (PCBs), DDE, hexachlorobenzene (HCB) and trans-nonachlor were the most commonly detected organic contaminants in dipper eggs and prey. PCB and organochlorine patterns were highly consistent across sites within the Georgia Basin and between prey and dipper eggs, suggesting common sources of atmospheric deposition to the region. Concentrations of total organochlorines and PCBs in dipper eggs were positively related to drainage area and collection year, but not to elevation. Total PCBs in dipper eggs and prey were dominated by congeners 153, 138 and 180. There were no differences in egg congener patterns between Chilliwack egg samples and other Georgia Basin samples. However, principle component analysis revealed significant spatial differences among dipper eggs collected from the main stem of the Chilliwack River and those from smaller tributaries, primarily due to variation in the lower chlorinated PCBs 66 and 105. Biomagnification of contaminants occurred between trophic levels for DDE, HCB, trans-nonachlor, and total PCBs with the highest biomagnification factors calculated for PCBs and DDE. This

study provides a benchmark for using a unique aquatic passerine, the American dipper, for biomonitoring persistent organic pollutants in mountain streams of western North America.

Key words: atmospheric deposition, American dipper, organochlorines, PCBs, cold condensation, biomonitor, biomagnification, salmon fry, benthic invertebrates, lotic waters

Introduction

Persistent and toxic substances from agricultural, urban and industrial processes are major sources of pollutants to remote regions world-wide. Semi-volatile compounds, such as organochlorine pesticides (OCs) and polychlorinated biphenyls (PCBs) have frequently been detected in remote northern latitudes, often exceeding levels found in more industrialized regions (Dewailly et al. 1989). Deposition of pollutants in distant regions is largely a result of atmospheric transport, precipitation and cold condensation- a process of volatilization in warmer low latitude regions and subsequent condensation in colder high latitudes (Wania and Mackay 1993). High elevation mountain ranges are similar to northern arctic environments in their low average temperatures and high precipitation, thus promoting atmospheric deposition from cold condensation (Blais et al. 1998). In fact, Davidson et al. (2003) demonstrated that terrestrial vegetation in alpine ecosystems could accumulate OCs and PCBs to the same degree as observed in polar regions.

Atmospherically deposited organic pollutants have contaminated freshwater aquatic ecosystems across North America (Swackhamer and Hites 1988, Johnson et al. 1988, Donald 1993, Datta 1998). In western Canada, rivers and lakes receiving annual snowmelt and glacial melt from surrounding mountains have been found to contain significant levels of persistent organic pollutants from atmospheric sources (Donald 1993, Blais et al. 2001). Therefore, watersheds can act as sinks for pesticides and PCBs from local and long-range sources. Because of their persistence and bioaccumulation potential, many organic compounds are rapidly assimilated in biological tissues and biomagnified in food chains to levels sometimes approaching or in excess of human

consumption guidelines (Kidd et al. 1995). In Canada and Europe, OC concentrations in fish increased with lake elevation indicating sources were primarily atmospheric in origin (Donald et al. 1998, Grimalt et al. 2001). While many previous studies have identified atmospheric deposition as a major contributor to food chain contamination in both arctic environments (Norstrom and Muir 1994, Bard 1999, Wilson et al. 1995) and remote lake ecosystems (Johnson et al 1988, Swackhamer and Hites 1988, Macdonald and Metcalfe 1991), the phenomenon has rarely been studied in mountainous lotic systems where deposition would be expected.

The effectiveness of birds as pollution monitors in a variety of ecosystems is widely established (Furness 1993, Burger 1993). However, few species of birds occupy mountain streams year-round or feed exclusively in lotic environments, thus reducing their value in identifying the specific sources of contaminants to watersheds. The American dipper (*Cinclus mexicanus*) demonstrates significant potential as an indicator of watershed pollution because of its year round residency in mountain streams and its exclusively aquatic diet of benthic invertebrates and small fish. Dippers are territorial and typically make only short altitudinal movements within a watershed (Morrissey et al. submitted- a). Although dippers in Europe (*Cinclus cinclus*) have been extensively used as monitors of organochlorines, PCBs and mercury pollution (Ormerod and Tyler 1990, 1992, O'Halloran et al. 1992, 2003, Ormerod et al. 2000), the North American species has not been studied for this purpose until recently.

The Fraser River Valley cuts through the Coast Mountains of southwestern British Columbia, Canada. Prevailing oceanic westerly winds carry pollutants originating from the large urban area of the city of Vancouver up the valley causing the Fraser Valley

to receive a combination of urban, suburban, marine, and agricultural emissions of pollutants. Thus, the Chilliwack River watershed, located in the central Fraser Valley was hypothesized to receive inputs of persistent organic pollutants through wet and dry deposition processes. Therefore, our main objective was to characterize the degree of contaminant contributions to this watershed's food chain from local and long-range atmospheric deposition. Given the relative lack of knowledge about the use of passerines for biomonitoring atmospherically deposited organic contaminants in aquatic environments, we attempted to estimate the degree to which those pollutants were able to biomagnify in lotic food chains using a unique aquatic songbird species and its diet. As such, this study is the first in North America to identify the patterns of organochlorine and PCB deposition to mountain streams using the eggs and prey of an obligate aquatic songbird, the American dipper.

Methods

Study area

The study area was located within the Chilliwack River watershed (49°1'N, 121°4'W) in the Georgia Basin, British Columbia, Canada (Fig. 1). The Chilliwack watershed is a tributary of the large Fraser River system, located approximately 100 km east of Vancouver in the Cascade Mountain Range and drains an area of 1274 km^2. Elevation ranges from near sea level on the Chilliwack-Vedder River to over 2000 m at several mountain peaks. The watershed tributaries are typically first through third order streams. The river is dominantly a fourth or fifth order stream, fed by a large glacial lake at its upper end and is 43.5 km in length where it merges into the Vedder River. Annual

precipitation to the Chilliwack area averages 1850.5 mm with mean daily temperatures of 10.4°C (Data from 1879-1990).

The Chilliwack River supports populations of Pacific chum, coho, pink, and chinook salmon (*Oncorhynchus keta, O. kisutch, O. gorbuscha, O. tshawytscha*) as well as cutthroat trout (*O.clarki*), steelhead trout (*O. mykiss*), and Dolly Varden (*Salvelinus malma*) (B.C. Fisheries data). Annual breeding runs of anadromous salmon and steelhead spawn from late summer through winter within the watershed, but peak runs occur along the main stem of the river and at the hatchery in autumn. Juvenile salmon are most abundant in spring and summer feeding on a variety of benthic aquatic invertebrate larvae present throughout the watershed.

Collection of aquatic invertebrates and fish

Composite samples of benthic invertebrates were collected at eight different sites along the main stem of the river and from seven different tributaries in the watershed in April 2002 prior to the spring freshet. Aquatic larval invertebrates (~ 4 g wet weight) were collected either by kick sampling in the stream (disturbing the rocks directly upstream of a Surber sampler) or by turning over rocks by hand. The sample represented a mixture of insect taxa that dippers would naturally prey upon including approximately equal proportions of Ephemeropteran, Plecopteran and Tricopteran larvae in addition to a much smaller fraction by mass of Coleopteran and Dipteran larvae. During invertebrate collections, eight additional composite samples consisting of ten individual salmon fry (*Oncorhynchus spp.*) (age-0) that each weighed 100-200 mg fresh weight, were captured live from the same eight sites along the main river using a dip net. Each collection represented a pooled sample of predominantly coho and chum salmon fry (~80%) but

pink and chinook salmon (~20%) were also included at some sites. All samples were subsequently washed three times with distilled deionized water to remove any surface contamination or stream water and stored frozen in acetone: hexane washed glass vials until preparation for contaminant residue analysis.

Egg collection and contaminant residue analysis

We collected a single egg from a clutch of four or five eggs at 32 dipper nests throughout the Chilliwack watershed during spring 1999, 2000 and 2001. In addition, a single dipper egg was collected at random from 3 other nests on rivers in the Georgia Basin of B.C. in 2000: the Coquitlam River, Seymour River and Cheakamus River (Fig. 1). Both fresh and failed eggs were used for the contaminant analysis. Fresh eggs were selected from the clutch at random during incubation while failed eggs caused by infertility, abandonment or embryo mortality were collected during routine banding of nestlings. There were no differences in contaminant burdens between fresh (viable) and failed eggs (total PCBs: $t_{30} = -0.13$, $p = 0.9$; total OCs: $t_{30} = -1.6$, $p = 0.1$) permitting the sample to be pooled for statistical analyses. Similarly, no differences in moisture or lipid content existed between fresh and failed eggs ($p > 0.05$); however, arithmetic corrections were applied to wet weight residues of any desiccated samples that deviated by more than 5 % from the mean moisture content (79 %) of freshly collected, undeveloped eggs. Whole eggs were stored refrigerated for up to four weeks. Egg contents were then transferred into an acetone: hexane rinsed glass jar to be frozen at –25°C until analysis.

Chemical analyses of all egg samples were carried out at the National Wildlife Research Centre (NWRC) in Hull, Quebec. Organochlorine analyses included determination of chlorobenzenes (tetrachlorobenzene, pentachlorobenzene,

hexachlorobenzes), hexachlorocyclohexanes (α-,β-,γ-HCH), chlordane related compounds (oxychlordane, trans-chlordane, cis-chlordane, trans-nonachlor, cis-nonachlor and heptachlor epoxide), DDT and metabolites (p,p'-DDE and p,p'-DDD), mirex and photomirex, and dieldrin. Total PCBs were calculated by summing the peaks of 62 individual congeners identified. Samples were quantitatively analyzed by capillary gas chromatography coupled with a mass selective detector operated in selected ion monitoring mode according to CWS Method No. MET-CHEM-OC-04B (Won et al. 2000). Briefly, samples underwent neutral extraction with 1:1 DCM/Hexane after dehydration with anhydrous sodium sulfate, removal of lipids and biogenic materials by Gel Permeation Chromatography, and further cleanup by Florisil column chromatography. All samples to be analyzed were spiked with internally labeled ^{13}C standards prior to extraction. Each sample extract was injected twice, once for determination of organochlorines and once for PCBs. As part of the quality control, blanks and CWS reference material (1989 Lake Ontario Herring Gull QA) were run concurrently with the samples. The nominal detection limit for all compounds was 0.0001 μg/g wet weight. Internal standard recoveries were typically between 80 % and 110 %. Residues were not corrected for internal standard recoveries and all egg values are reported on a wet weight basis (Peakall and Gilman 1979).

Data analysis

Direct measurements of elevation (± 1 m) for each nest site in the watershed were taken using a calibrated digital altimeter (Suunto Escape 203, Finland). Environment Canada and U.S. Geological Survey hydrological databases provided digital watershed maps to calculate the drainage area for each breeding territory where eggs were collected.

Chemical residue data was log normally distributed (KSL test) and subsequently $\log_{10}$ transformed prior to statistical analyses. We report geometric mean values and data ranges. Where levels were below the detection limit for organochlorine compounds, a value equal to the minimum detection limit (0.0001 μg/g wet weight) was applied.

The process of atmospheric deposition to watersheds was hypothesized to affect differences in levels of contaminants between invertebrate prey from the river and invertebrates from the tributaries. Therefore, a one-way ANOVA was used to compare contaminant concentrations among prey types (river invertebrates, creek invertebrates and salmon fry). A two-sample t-test was used to compare egg concentrations between dipper egg sampling locations (Chilliwack vs. other Georgia Basin sites). We hypothesized that drainage area and elevation would be significant predictors of contaminant levels in dipper eggs if atmospheric deposition was occurring. Therefore, multiple regression models for predicting concentrations of total OCs and total PCBs included elevation, basin area, year of collection and significant interaction terms. Selected PCB congeners that contributed greater than 1 % to the total PCBs were used to identify trends in dipper eggs and prey. PCB congener patterns in dipper eggs were identified by principal component analysis (PCA). Eggs were grouped by breeding location (Georgia Basin, Chilliwack river and Chilliwack creeks) to determine if any differences in congener contributions were occurring. All statistical analyses were performed using JMP IN v. 4.0.

Biomagnification factors (BMFs) were calculated on a wet weight basis for only the most commonly detected compounds (DDE, HCB, trans-nonachlor, total PCBs). We divided the geometric mean concentration in dipper eggs by the geometric mean

concentration in the diet; assuming average dipper diet was comprised of 67% invertebrates and 33 % fish (Morrissey et al. submitted-b).

Results

DDE, hexachlorobenzene (HCB) and trans-nonachlor were the only organochlorine compounds routinely detected in benthic invertebrates and salmon fry collected from the Chilliwack River watershed (Table 1). PCBs were not detected in any of the invertebrate samples (n = 15) and only half of the salmon fry samples (n = 8). Higher concentrations of DDE occurred in salmon fry relative to invertebrates collected from both the main river and the tributaries ($F_{2,20} = 9.02$, $p < 0.002$). Trans-nonachlor was also higher in fish and river invertebrates compared to creek invertebrates ($F_{2,20} = 7.88$, $p = 0.003$), however, no significant differences in HCB levels existed among prey types.

Eggs from Chilliwack dippers showed very similar patterns to their prey with respect to detection of common contaminants (Table 2). DDE, PCBs and HCB were detected in 100% of the egg samples and at the highest concentrations, while trans-nonachlor was detected in 75% of the samples and represented a significant proportion of total organochlorines. All other OCs were detected at a much lower frequency and in very low concentrations (other chlorobenzenes and chlordanes, hexachlorocyclohexanes, DDT and DDD). Dieldrin, mirex and photomirex were not detected in any egg samples. Dipper eggs from other sites in the Georgia Basin region showed a similar composition of organochlorine compounds as the Chilliwack samples (Table 2). Mean concentrations of common contaminants including total PCBs, DDE and trans-nonachlor did not differ between Chilliwack and the other 3 Georgia Basin eggs combined. However, levels of

HCB were significantly lower at other Georgia Basin sites compared to Chilliwack primarily because of the low concentration in the Coquitlam River egg sample ($t_{33} = 2.6$, $p = 0.01$).

Concentrations of total OCs and total PCBs found in American dipper eggs were significantly related to the size of the drainage area where the egg was collected and the year of collection (total OCs: $F_{5, 26} = 8.82$, $p < 0.0001$, total PCBs: $F_{5,26} = 5.46$, $p = 0.002$). There was a positive trend towards increasing contaminant concentrations with increasing drainage area, which was stronger for total OCs than total PCBs (Fig. 2a and 2b). Although elevation and drainage area were negatively correlated ($r = - 0.59$, $p = 0.0003$), only drainage area was a significant predictor of total OCs ($p = 0.002$) and total PCBs ($p = 0.03$). For both the total OC and total PCB regression models, contaminant levels were significantly higher in 1999 than in the 2 subsequent collection years, 2000 and 2001 (total OCs: $p = 0.002$, total PCBs: $p = 0.004$).

Concentrations of total PCBs, DDE, HCB and trans-nonachlor increased with each trophic level (Fig. 3). Similar lipid contents were found between prey types: benthic invertebrates (3.1 %) and salmon fry (2.4 %). Total PCBs and DDE contributed to the largest increases between trophic levels. As a result, BMFs calculated from the dipper's diet (assuming diet of 67 % invertebrates and 33 % fish) to eggs was highest for total PCBs and DDE (Table 1). Since salmon fry were the only diet items found with detectable levels of PCBs, they were likely the main source of PCBs to dippers. Therefore, we calculated an alternate BMF of 87 from salmon fry to dipper eggs. HCB and trans-nonachlor were also observed to biomagnify from dipper prey to eggs since BMF values were greater than 1. BMFs for other OC compounds were not reported

because of the low frequency of detection in prey or egg samples that prevented reliable estimates.

The percent contribution of individual PCB congeners was determined for Chilliwack dipper eggs, other Georgia Basin dipper eggs, and salmon fry from the Chilliwack River (Fig. 4). Congeners 153, 138, 180 and 118 dominated the PCB signature in Chilliwack dipper eggs, which together contributed 53 % to the total PCBs measured. Patterns were very similar for other Georgia Basin dipper eggs, which were dominated by congeners 153, 138, 180 and 170/190. Furthermore, PCB congeners detected in juvenile salmon closely modeled those in the eggs of its dipper predator, with major PCB congeners 153, 138, 180 and 187 making up 51 % of the total.

PCA analysis on 18 congeners that contributed > 1 % to total PCBs, revealed two significant principle components that explained 82 % of the variation in Chilliwack dipper eggs. The first component (PC1) explained 71 % of the variation while the second component (PC2) explained 11 %. Congener patterns were not significantly different for Chilliwack eggs and other Georgia Basin sites indicating common atmospheric sources (Fig. 5). However, dippers occupying the main stem of the Chilliwack River had a different congener pattern than dippers breeding on tributaries. Grouping of creek dippers separate from river birds was primarily attributed to variation on PC2 in the lower chlorinated PCB 66 and PCB 105 (Table 3). Creek eggs had significantly higher scores on that component than river eggs ($t_{27} = 3.5$, $p = 0.002$). Increasing hexa and hepta-chlorinated biphenyls (PCB 180, 146 and 183) on PC1 produced higher scores for river eggs on this component, thus further contributing to observed differences in congener patterns ($t_{27} = -2.5$, $p = 0.02$).

Discussion

In general, levels of many persistent organic pollutants in American dipper eggs and prey were very low, often at or below levels of detection. However, the consistent presence and uniformity of 3 major organochlorines (DDE, HCB and trans-nonachlor) and common PCB congeners in prey samples and dipper eggs collected from all four sites suggests that sources of organochlorines to the Georgia Basin region appear to be largely atmospheric in origin. This is consistent with the study by Elliott et al. (1989a) who suggested that OC contaminants in seabird eggs were primarily from atmospheric sources that dominated a wide area of the British Columbia coast. Elliott et al. (1996) also detected similar OC and PCB profiles in Bald eagle (*Haliaeetus leucocephalus*) eggs from the Georgia Basin region, with dominant compounds PCBs, DDE and trans-nonachlor present in the highest concentrations. These common organic compounds were either generally released in this area or were transported from distant sources and dispersed widely throughout the region.

The Fraser Valley is an intensively farmed region of British Columbia where organochlorine pesticides were widely applied to crops prior to the mid-1970s. Volatilization of persistent residues from soils in the Fraser Valley remains an important contributor of organochlorine pesticides to the atmosphere decades after their ban (Szeto and Price 1991, Falconer et al. 1997). Consistent with Chilliwack's location in the central Fraser Valley, DDE levels appeared higher in the Chilliwack dipper eggs compared to other sites, which is likely due to extensive historic use of the parent compound DDT (Finizio et al. 1998). HCB had greater variability among sampling sites with higher concentrations in eggs from the Cheakamus River and Chilliwack River. A

minimum detectable concentration of HCB found in the Coquitlam sample caused the Chilliwack mean to appear higher than other Georgia Basin sites. Since this compound has several origins as a fungicide and an industrial waste product from production of chlorinated solvents and pesticides (Courtney 1979), its points of release may be patchy, causing similar trends in deposition. In general, differences in residue concentrations among sampling sites in the Georgia Basin were likely a product of variability in atmospheric deposition and the low sample size outside the Chilliwack watershed.

We hypothesized that concentrations of persistent organic pollutants would be related to elevation or drainage area if atmospheric deposition were occurring. Snow pack samples and fish collected from alpine lakes have shown that higher elevation samples have greater concentrations of OCs and PCBs particularly the more volatile compounds (Donald 1993, Blais et al. 1998, Grimalt 2001). Furthermore, larger drainage areas should have cumulatively more input sources, resulting in higher contaminant loads. We found few differences in the concentrations of contaminants in invertebrates collected from the lower elevation, larger drainage area river sites compared to the higher elevation, smaller drainage area tributaries. DDE appeared higher in river invertebrate samples compared to creek samples, but only trans-nonachlor was significantly higher. Increasing elevation did not have an additional influence on contaminant levels in dipper eggs after correcting for drainage area effects. Instead, drainage area and year were the only significant variables that explained variation in contaminant levels of dipper eggs. It is possible that our range of elevations where eggs were collected was not great enough to detect a difference (32 - 600 m). Alternatively, for lotic environments, elevation may not be a significant factor because the main site of

deposition is in high elevation snow pack and precipitation, which exposes birds and prey regardless of elevation as contaminated waters progress downstream. Inputs from several smaller tributaries merge at downstream locations causing increases in residue levels from cumulative sources, as were revealed by analysis of eggs but only marginally in the invertebrate prey. The year of collection was clearly an important factor in explaining variation in contaminant burdens to eggs. In particular, 1999 had higher concentrations of OCs and PCBs than the two subsequent years combined. Large increases in snowpack loadings of less volatile compounds have been attributed to higher precipitation and snow depth (Blais 1998). Since the winter of 1998-1999 had record snowfall levels throughout the region, this likely increased the concentrations of contaminants in runoff for that year.

Similar compositions of organic contaminants were found in both invertebrates and fish but residues in salmon fry tended to exceed those of benthic invertebrates. Dippers are therefore, likely to receive the majority of their OC and PCB burdens from ingestion of salmon fry. PCBs were detected in fish samples but not in any of the composite invertebrate samples from the Chilliwack watershed, while DDE and trans-nonachlor were also significantly higher in fish over invertebrates. The occurrence of elevated residue levels of various xenobiotics in aquatic biota with increasing trophic levels has been attributed to food chain biomagnification (Oliver and Niimi 1988, Kidd et al. 1995). However, LeBlanc (1995) found that trophic level differences are largely due to increased bioconcentration because of higher lipid content of upper trophic level organisms. Our results demonstrate that biomagnification was the major underlying process affecting differences in contaminant residues between fish and invertebrates, since lipid contents were similar or higher in the lower trophic level invertebrates.

Information on the biomagnification potential of low-level atmospherically deposited contaminants to lotic ecosystems has not been previously reported. BMFs offer the potential to estimate avian egg concentrations just from analyzing diet items, assuming these items are the predominant food eaten and differences in metabolism between avian species are not too large (Hoffman et al. 1996). Since reproductive dysfunction is generally the most critical and sensitive endpoint for population studies, knowledge of egg residues is an important predictor of potential reproductive effects (Hoffman et al. 1996). In this study, BMFs from American dipper prey to eggs were highest for total PCBs and DDE. Harris et al. (2000) recorded DDE bioaccumulation factors of 14 to 61 (earthworms to American robin (*Turdus migratorius*) eggs) in orchard food chains, which encompassed our value for dippers (31). Braune and Norstrom (1989) also provided data on BMFs in the Great Lakes food chain from fish prey to Herring gull (*Larus argentatus*) eggs. Although their published BMF value for DDE of 34 closely matches our value of 31, total PCBs biomagnified 32x from alewife to Herring gull eggs, which was considerably lower than our estimate of 175 from dipper prey to eggs. Henny et al. (2003) also reported a much lower BMF value for total PCBs (11) in fish to Osprey (*Pandion haliaetus*) eggs. This discrepancy may be attributed to dippers accumulating greater concentrations of PCBs from their prey, greater excretion of PCBs into eggs which reduces the female body burden, differences in lipid content of eggs, or large variability in prey and egg values by watershed location. Alternatively, our method for calculating BMFs from the diet apportioning may have been biased, since PCBs were only detected in salmon fry. When only salmon fry are used in the calculation, the BMF value is reduced to 87. The method of estimating BMFs is designed to predict

contaminant burdens to wildlife consumers and estimate the potential risk of exposure prior to analyzing eggs. Therefore, we need to improve our knowledge of BMF estimates from a wide range of species with different diets that are accurately identified to ensure reliable predictions of egg residues.

PCB congeners detected in this study are frequent and widespread components of common commercial PCB mixtures. PCB 153, 138 and 180 dominated the contribution to total PCBs in dipper eggs and salmon fry. Elevated levels of PCBs 153, 138 and 180 have been associated with the presence of Aroclor 1260, a common constituent in transformer and capacitor oils, because of their high proportions in this mixture (Jones 1988). Other avian species occupying different ecosystems throughout this region are reported to have those congeners in higher concentrations within their eggs (Elliott et al. 1989b, Elliott et al. 1996), indicating widespread use of the mixture and persistence of those specific congeners surviving atmospheric transport processes.

Although we found no differences in PCB congener patterns among Chilliwack dipper eggs and other Georgia Basin dipper eggs, PCA analysis revealed that congener composition in eggs collected from the main river and the tributaries of the Chilliwack watershed were significantly separated. Much of the distinction between river and creek dippers was attributed to variation in the lower chlorinated PCBs 66 and 105. Creek birds had higher scores on PC2 representing greater contributions of those congeners, whereas river birds had higher scores on PC1 for the more highly chlorinated and stable PCB congeners. Blais et al. (1998) found that the more volatile, less chlorinated PCBs were prevalent in higher elevation snow pack. Creek eggs were collected from territories at higher average elevations (creek: 328 ± 43 vs. river: 208 ± 25) with smaller drainage

areas (creek: 96 ± 12 vs. river: 615 ± 40) than river birds, possibly explaining the greater presence of lower chlorinated congeners in eggs. Longitudinal patterns have been shown to exist in watersheds with respect to conventional changes that occur in stream flow, water chemistry, nutrient levels, and contaminant loads from headwaters to downstream river sites (Giller and Malmqvist 2000). Therefore, PCB congener patterns may also follow a similar trend in biota. Alternatively, data by Morrissey et al. (submitted- b) showed that river resident dippers ingest larger proportions of salmon fry than creek migrants, so that differences in congener patterns may be attributed to differences in the diet.

Atmospheric deposition in precipitation appeared to be the major source of chlorinated hydrocarbons to the Chilliwack watershed; however, other non-point sources may have amplified the residue levels found in lotic biota. For example, biotransport of organic pollutants by anadromous fish can represent a significant source of contamination to streams and rivers (Merna 1986, Giesy et al 1994, Ewald et al. 1998, Krummel et al. 2003). Large annual salmon and steelhead runs on the Chilliwack River contribute substantial organic matter to the lotic food chain via salmon roe and decaying carcasses (Kline 1993). Decaying salmon may have increased the organic contaminant loads in peak spawning areas, particularly along the main river channel. However, contributions from salmon are probably minor since we found few differences in contaminant concentrations in benthic invertebrates collected from the main river where salmon densities are high, and the tributaries, where salmon densities are lower (B.C. Fisheries data). Other sources of OCs and PCBs may have come from commercial salmon feed released from the local hatchery (Easton et al. 2002). Salmon feed is incorporated into

the lotic food chain through ingestion by hatchery juvenile salmon or benthic invertebrates in and around the hatchery waterways, which are routinely preyed upon by dippers. Future research investigating contaminant burdens in dippers occupying streams with and without anadromous fish and hatchery inputs would be valuable to compare levels of OCs and PCBs in lotic food chains.

Levels of OCs and PCBs measured in this study generally fell within or below the range of previously reported concentrations in other passerine eggs from British Columbia and in dipper eggs from Europe (*Cinclus cinclus*) (Table 4). Egg residues of dippers exposed through atmospheric deposition in the Chilliwack River watershed, were similar to those of other species in orchard and non-orchard terrestrial habitats in British Columbia (Elliott et al. 1994). None of those species, including American robins that were more heavily contaminated by DDE, showed any measurable adverse effects on reproductive performance. Studies on dippers in Europe also found no strong evidence of reproductive effects or reduced post-fledging survival with OC and PCB concentrations equal to or in excess of those detected in this study (Ormerod and Tyler 1992, Ormerod et al. 2000) (Table 4). Dipper eggs in this study had mean total PCB and DDE concentrations less than 0.1 ppm, and all individual eggs were below 1 ppm wet weight. The lowest observable effect level for total PCBs in chickens, the most sensitive avian species, is 1 to 5 ppm in eggs (Hoffman et al. 1996). For DDE, the Brown pelican (*Pelecanus occidentalis*) is the most sensitive species, demonstrating eggshell thinning and decreased productivity at 3 ppm in eggs (Blus 1996). Subtle sublethal effects, including physiological changes, can occur as a result of in ovo exposure to concentrations lower than those associated with overt reproductive effects (Hoffman et al.

1996). However, we have no evidence from published literature to suggest that the current levels of atmospheric deposition and biomagnification of organic pollutants in Georgia Basin watersheds are putting dipper populations at risk.

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Literature Cited

Bard, S.M. 1999. Global transport of anthropogenic contaminants and the consequences for the Arctic marine ecosystem. Mar. Pollut Bull. 38: 356-379.

Blais, J.M., Schindler, D.W., Muir, D.C.G., Kimpe, L.E., Donald, D.B., and Rosenberg, B. 1998. Accumulation of persistent organochlorine compounds in mountains of western Canada. Nature 395: 685-688.

Blais, J.M., Schindler, D.W., Muir, D.C.G., Sharp, M., Donald, D., Lafreniere, M., Braekevelt, E., and Strachan, W.M.J. 2001. Melting glaciers: a major source of persistent

organochlorines to subalpine Bow Lake in Banff National Park, Canada. Ambio 30: 410-415.

Blus, L.J. 1996. DDT, DDD, and DDE in birds. In Environmental contaminants in wildlife: Interpreting tissue concentrations. W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood (eds.). Setac Special Publication, CRC Press, Boca Raton.

Braune, B.M. and Norstrom, R.J. 1989. Dynamics of organochlorine compounds in herring gulls: III. Tissue distribution and bioaccumulation in Lake Ontario gulls. Environ. Toxicol. Chem. 8: 957-968.

Burger, J. 1993. Metals in avian feathers: bioindicators of environmental pollution. Rev. Environ. Toxicol. 5: 203-311.

Courtney, K.D. 1979. Hexachlorobenzene (HCB): a review. Environ. Res. 20: 225-266.

Datta, S., McConnell. L.L., Baker, J.E., Lenoir, J., and Seiber, J.N. 1998. Evidence for atmospheric transport and deposition of polychlorinated biphenyls to the Lake Tahoe Basin, California- Nevada. Environ. Scie. Technol. 32: 1278-1385.

Davidson, D.A., Wilkinson, A.C., Blais, J.M., Kimpe, L.E., McDonald, K.M., and Schindler, D.W. 2003. Orographic cold-trapping of persistent organic pollutants by vegetation in mountains of western Canada. Environ. Sci. Technol. 37: 209-215.

Dewailly, E.A., Nantel, J.P., Weber, J.P., and Meyer, F. 1989. High levels of PCBs in breast milk of Inuit women from Arctic Quebec. Bull. Environ. Contam. Toxicol. 43: 641-646.

Donald, D. 1993. Polychlorinated biphenyls and organochlorine pesticides in the aquatic environment along the continental divide region of Alberta and British Columbia. Inland Waters Directorate, Regina, Saskatchewan, Canada.

Donald, D.B., Stern, G.A., Muir, D.C.G., Fowler, B.R., Miskimmin, B.M., and Bailey, R. 1998. Chloroboronanes in water, sediment, and fish from toxaphene treated and untreated lakes in western Canada. Environ. Sci. Technol. 32: 1391-1397.

Easton, M.D.L., Luszniak, D., and Von der Geest, E. 2002. Preliminary examination of contaminant loadings in farmed salmon, wild salmon and commercial salmon feed. Chemosphere 46: 1053-1074.

Elliott, J.E., Noble, D.G., Norstrom, R.J., and Whitehead, P.E. 1989a. Organochlorine contaminants in seabirds from the Pacific coast of Canada, 1971-1986.

Elliott, J.E., Butler, R.W., Norstrom, R.J., and Whitehead, P.E. 1989b. Environmental contaminants and reproductive success of Great Blue herons *Ardea herodias* in British Columbia, 1986-87. Environ. Pollut. 59: 91-114.

Elliott, J.E., Marin, P.A., Arnold, T.W., and Sinclair, P.H. 1994. Organochlorines and reproductive success of birds in orchard and non-orchard areas of central British Columbia, Canada, 1990-91. Arch. Environ. Contam. Toxicol. 26: 435-443.

Elliott, J.E., Norstrom, R.J., and Smith, G.E.J. 1996. Patterns, trends, toxicological significance of chlorinated hydrocarbon and mercury contamination in Bald Eagle eggs from the Pacific coast of Canada, 1990-1994. Arch. Environ. Contam. Toxicol. 31: 354-367.

Ewald, G., Larsson, P., Linge, H., Okla, L., and Szarzi, N. 1998. Biotransport of organic pollutants to an inland Alaska lake by migrating Sockeye salmon (*Oncorhynchus nerka*). Arctic 51: 40-47.

Falconer, R.L., Bidleman, T.F., and Szeto, S.Y. Chiral pesticides in soils of the Fraser Valley, British Columbia. J. Agric. Food Chem. 45: 1946-1951.

Finizio, A., Bidleman, T.F., and Szeto, S.Y. 1998. Emission of chiral pesticides from an agricultural soil in the Fraser Valley, British Columbia. Chemosphere 36: 345-355.

Furness, R.W. 1993. Birds as monitors of pollutants. In Birds as Monitors of Environmental Change (eds. R.W. Furness and J.J. D. Greenwood). Chapman and Hall, London, pp. 86-143.

Geisy, J.P., Verbrugge, D.A., Othout, R.A., Bowerman, W.W., Mora, M.A., Jones, P.D., Newsted, J.L., Vandervoort, C., Heaton, S.N., Aulerich, R.J., Bursian, S.J., Ludwig, J.P., Ludwig, M., Dawson, G.A., Kubiak, T.J., Best, D.A., and Tillit, D.E. 1994. Contaminants in fishes from Great Lakes-influenced sections and above dams of three Michigan rivers. I: Concentrations of organochlorine insecticides, polychlorinated biphenyls, dioxin equivalents, and mercury. Arch. Environ. Contam. Toxicol. 27: 202-212.

Giller, P.S. and Malmqvist, B. 2000. The Biology of Streams and Rivers. Oxford University Press, New York.

Grimalt, J.O., Fernandez, P., Berdie, L., Vilanova, R.M., Catalan, J., Psenner, R., Hofer, R., Appleby, P.G., Rossland, B.O., Massabuau, J.C., and Battarbee, R.W. 2001. Selective trapping of organochlorine compounds in mountain lakes of temperate areas. Environ. Sci. Technol. 35: 2690-2697.

Harris, M.L., Wilson, L.K., Elliott, J.E., Bishop, C.A., Tomlin, A.D., and Henning, K.V. 2000. Transfer of DDT and metabolites from fruit orchard soils to American Robins (*Turdus migratorius*) twenty years after agricultural use of DDT in Canada. Arch. Environ. Contam. Toxicol. 39: 205-220.

Henny, C.J., Kaiser, J.L., Grove, R.A., Raymond Bentley, V., and Elliott, J.E. 2003. Biomagnification factors (fish to Osprey eggs from Willamette River, Oregon, U.S.A.) for PCDDs, PCDFs, PCBs and OC pesticides. Environ. Monitor. Assess. 84: 275-315.

Hoffman, D.J., Rice, C.P., and Kubiak, T.J. 1996. PCBs and dioxins in birds. In Environmental contaminants in wildlife: Interpreting tissue concentrations. W.N. Beyer, G.H. Heinz and A.W. Redmon- Norwood (eds.). Setac Special Publication, CRC Press, Boca Raton.

Johnson, M.G., Kelso, J.R.M., and George, S.E. 1988. Loadings of organochlorine contaminants and trace elements to two Ontario lake systems and their concentrations in fish. Can. J. Fish. Aquat. Sci. 45 (Suppl. 1): 170-178.

Jones, K.C. 1988. Determination of polychlorinated biphenyls in human foodstuffs and tissues: suggestions for a selective congener analytical approach. Sci. Tot. Environ. 68: 141-159.

Kallenborn, R., Planting, S., Haugen, J.-E., and Nybo, S. 1998. Congener-, isomer- and enantiomer-specific distributionof organochlorines in dippers (*Cinclus cinclus L.*) from southern Norway. Chemosphere 37: 2489-2499.

Kidd, K.A., Schindler, D.W., Muir, D.C.G., Lockhart, W.L., and Hesslein, R.H. 1995. High concentrations of toxaphene in fishes from a subarctic lake. Science 269: 240-242.

Kline, T.C. Jr., Goering, J.J., Mathisen, O.A., Poe, P.H., Parker, P.L., and Scalan, R.S. 1993. Recycling of elements transported upstream by runs of Pacific salmon: II. δ15N

and δ13C evidence in the Kvichak River watershed, Bristol Bay, Southwestern Alaska. Can. J. Fish. Aquat. Sci. 50: 2350-2365.

Krummel, E.M, Macdonald, R.W., Kimpe, L.E., Gregory-Eaves, I., Demers, M.J., Smol, J.P., Finney, B. and Blais, J.M. Delivery of pollutants by spawning salmon. Nature 425: 255-256.

LeBlanc, G.A. 1995. Trophic-level differences in the bioconcentrationof chemicals: implications in assessing environmental biomagnification. Environ. Sci. Technol. 29: 154-160.

Macdonald, C.R. and Metcalfe, C.D. 1991. Concentration and distribution of PCB congeners in isolated Ontario lakes contaminated by atmospheric deposition. Can. J. Fish. Aquat. Sci. 48: 371-381.

Merna, J.W. 1986. Contamination of stream fishes with chlorinated hydrocarbons from eggs of Great Lakes salmon. Trans. Amer. Fish. Soc.115: 69-74.

Morrissey, C.A., Bendell-Young, L.I., and Elliott, J.E. (submitted- a). Seasonal trends in population density, distribution and movement of American dippers within a watershed of southwestern British Columbia, Canada. Condor.

Morrissey, C.A., Bendell-Young, L.I., and Elliott, J.E. (submitted- b). Linking contaminant profiles to the diet and breeding location of American dippers using stable isotope analysis. J. Appl. Ecol.

Norstrom, R.J., and Muir, D.C.G. 1994. Chlorinated- hydrocarbon contaminants in arctic marine mammals. Sci. Total Environ. 154: 107-128.

O'Halloran, J., Ormerod, S.J., Smiddy, P., and O'Mahony, B. 1993. Organochlorine and mercury content of dipper eggs in south-west Ireland. Biol. and Environ. 93B: 25-31.

Oliver, B.G. and Niimi, A.J. 1988. Trophodynamic analysis of polychlorinated biphenyl congeners and other chlorinated hydrocarbons in the Lake Ontario ecosystem. Environ. Sci. Technol. 22: 388-397.

Ormerod, S.J. and Tyler, S.J. 1990. Environmental pollutants in the eggs of Welsh Dippers *Cinclus cinclus*: a potential monitor of organochlorine and mercury contamination in upland rivers. Bird Study 37: 171-176.

Ormerod, S.J., and Tyler, S.J. 1992. Patterns of contamination by organochlorines and mercury in the eggs of two river passerines in Britain and Ireland with reference to individual PCB congeners. Environ. Pollut. 76: 233-243.

Ormerod, S.J., Tyler, S.J., and Jüttner, I. 2000. Effects of point-source PCB contamination on breeding performance and post-fledging survival in the dipper *Cinclus cinclus*. Environ. Pollut. 109: 505-513.

O'Halloran, J., Irwin, S., Harrison, S., Smiddy, P. and O'Mahony, B. 2003. Mercury and organochlorine content of Dipper *Cinclus cinclus* eggs in south-west Ireland: trends during 1990-1999. Environ. Pollut. 123: 85-93.

Peakall, D.B. and Gilman, A.P. 1979. Limitations of expressing organochlorine levels in eggs on a lipid-weight basis. Bull. Environ. Contam. Toxicol. 23: 287-290.

Swackhamer, D.L. and Hites, R.A. 1988. Occurrence and bioaccumulation of organochlorine compounds in fishes from Siskiwit Lake, Isle Royale, Lake Superior. Environ. Sci. Technol. 20: 879-883.

Szeto, S.Y. and Price, P, M. 1991. Persistence of pesticide residues in mineral and organic soils in the Fraser Valley of British Columbia. J. Agric. Food Chem. 39: 1679-1684.

Wania, F., and Mackay, D. 1993. Global fractionation and cold condensation of low valatility organochlorine compounds in polar regions. Ambio 22: 10-18.

Wilson, R., Allengil, S., Griffin, D., and Landers, D. 1995. Organochlorine contaminants in fish from an arctic lake in Alaska, USA. Sci. Total Environ. 161: 511-519.

Won, H.T., Mulvihill, M.J., and Wakeford, B.J. 2000. Multi residue methods for the determination of chlorinated pesticides and polychlorinated biphenyls (PCBs) in wildlife tissues by gas chromatography/mass spectrometry. Technical Report Series No. 335. Canadian Wildlife Service, Environment Canada, Hull, Quebec.

Figure Legends

Figure 4.1. Map of the Georgia Basin region in southwestern British Columbia, Canada. Chilliwack River is shown relative to the city of Vancouver in addition to other American dipper egg collection sites: the Cheakamus, Seymour and Coquitlam Rivers.

Figure 4.2. Relationship between the size of the drainage area and a) total organochlorine or b) total PCB concentrations in American dipper eggs collected from sites within the Chilliwack River watershed, B.C. for 1999 and 2000/2001. Regression lines represent best fit for all data combined.

Figure 4.3. Relative concentrations (geometric means) of commonly detected organic contaminants in benthic invertebrates, juvenile salmon and American dipper eggs from the Chilliwack River watershed, B.C.

Figure 4.4. Percentage contributions by different congeners to the total PCB burden of American dipper eggs (n = 32) and salmon fry prey (n = 8) collected during 1999-2002 from the Chilliwack River watershed. Dipper eggs (n = 3) from 3 other sites in the Georgia Basin of British Columbia are included for comparison.

Figure 4.5. Principle component analysis of 18 selected PCB congeners that contributed greater than 1% to the total burden in American dipper eggs. Analysis shows principle component scores (PC1 and PC 2) for eggs grouped by collection site on the Chilliwack river, Chilliwack creeks and 3 other rivers in the Georgia Basin, B.C., 1999-2001.

gure 4.1

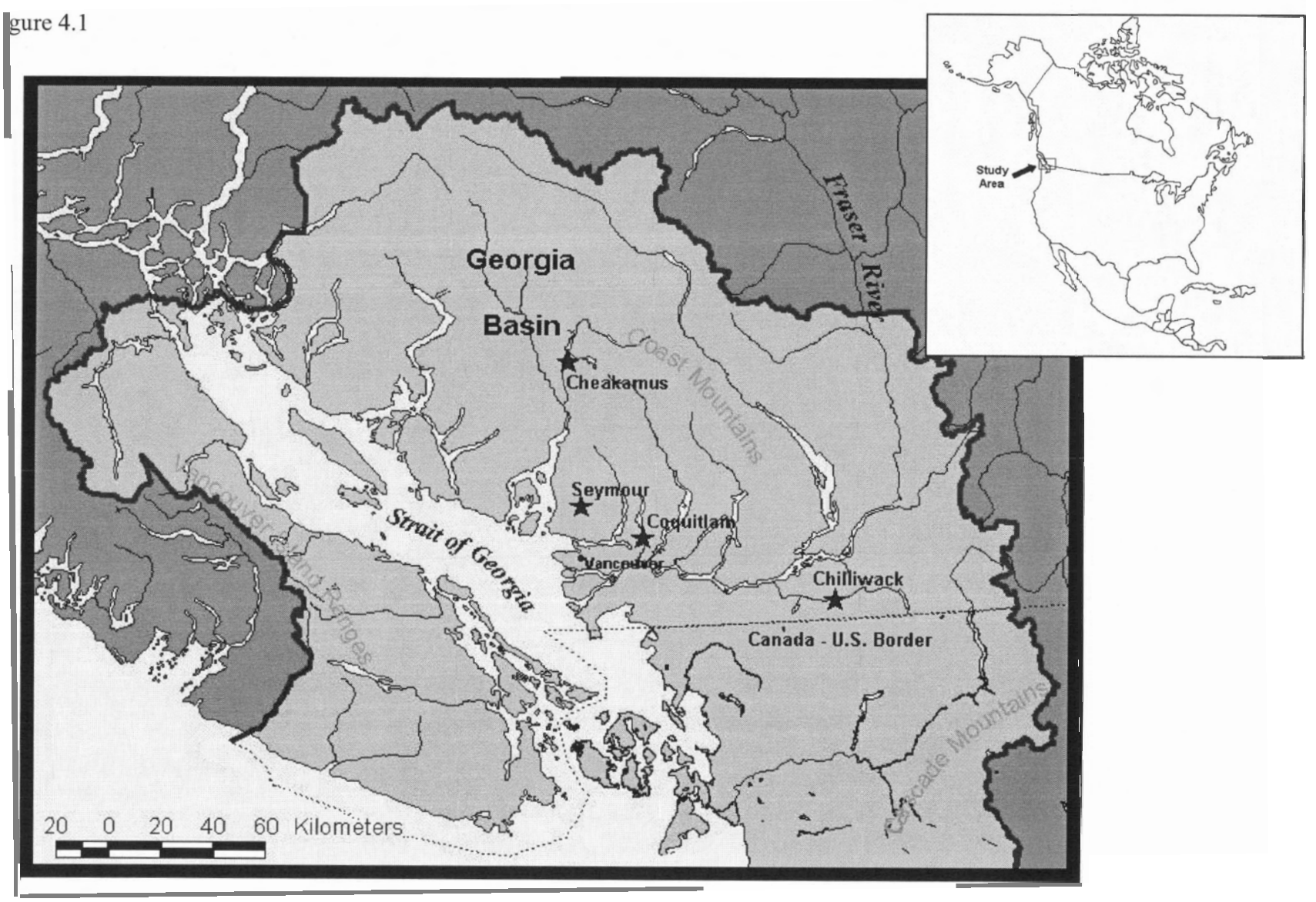
Georgia
Basin
Fraser River
Coast Mountains
Cheakamus
Seymour
Coquitlam
Vancouver
Strait of Georgia
Vancouver Island Ranges
Chilliwack
Canada - U.S. Border
Cascade Mountains
20 0 20 40 60 Kilometers
Study Area

Figure 4.2a

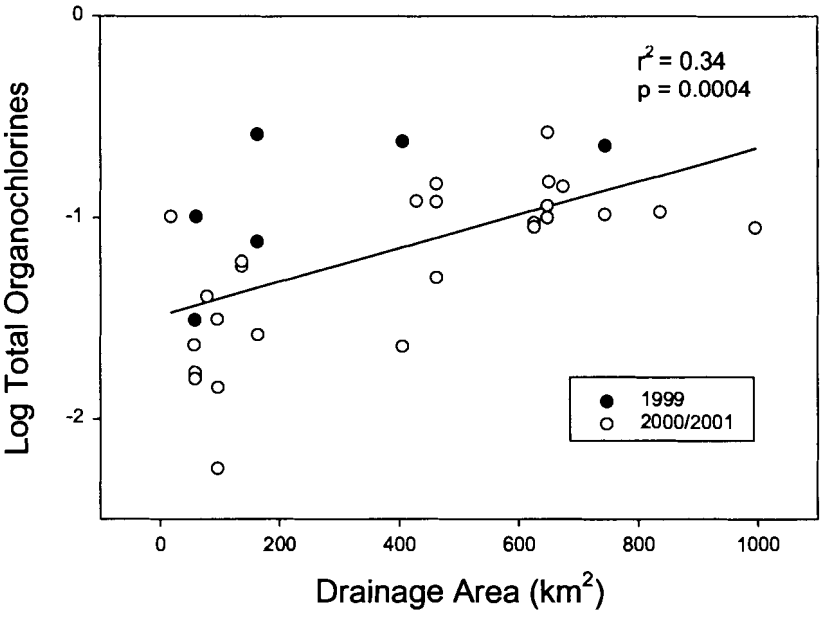

Figure 4.2b

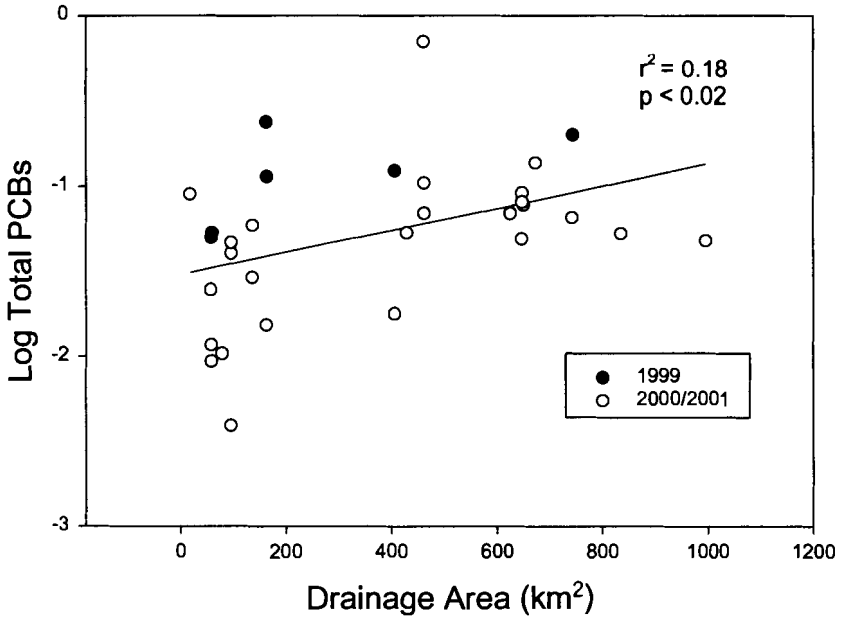

Figure 4.3

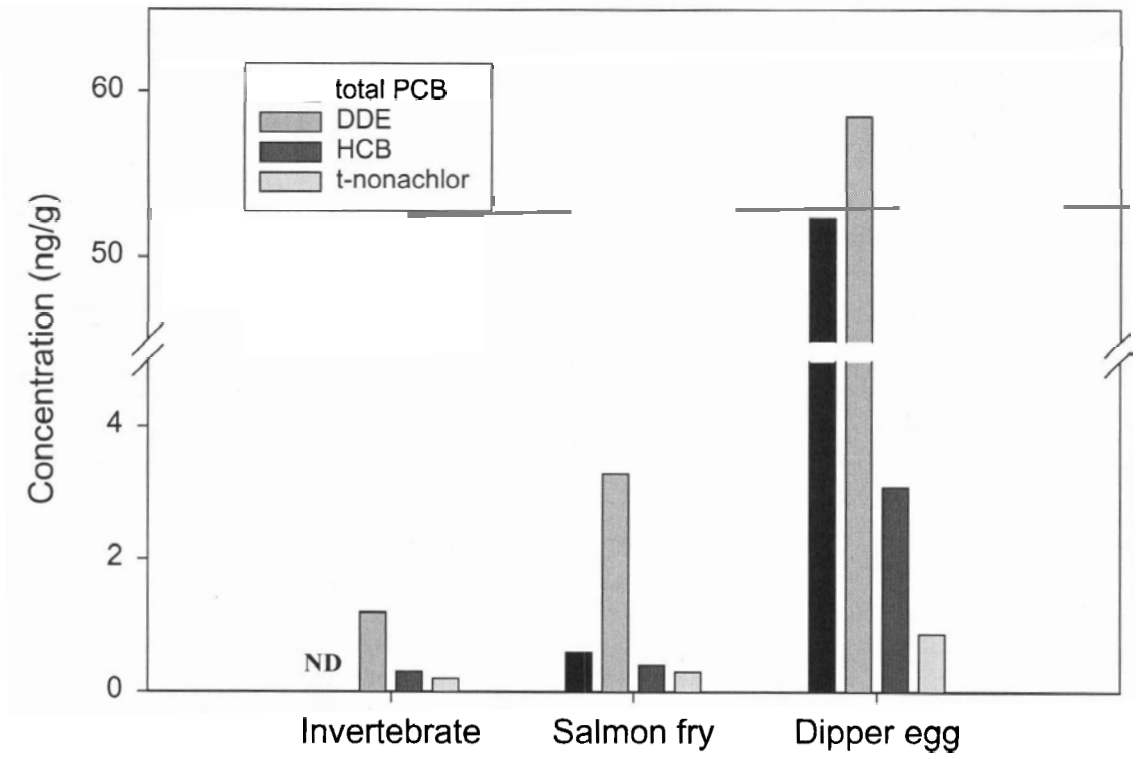
total PCB
DDE
HCB
t-nonachlor
Concentration (ng/g)
60
50
4
2
0
ND
Invertebrate
Salmon fry
Dipper egg

Figure 4.4

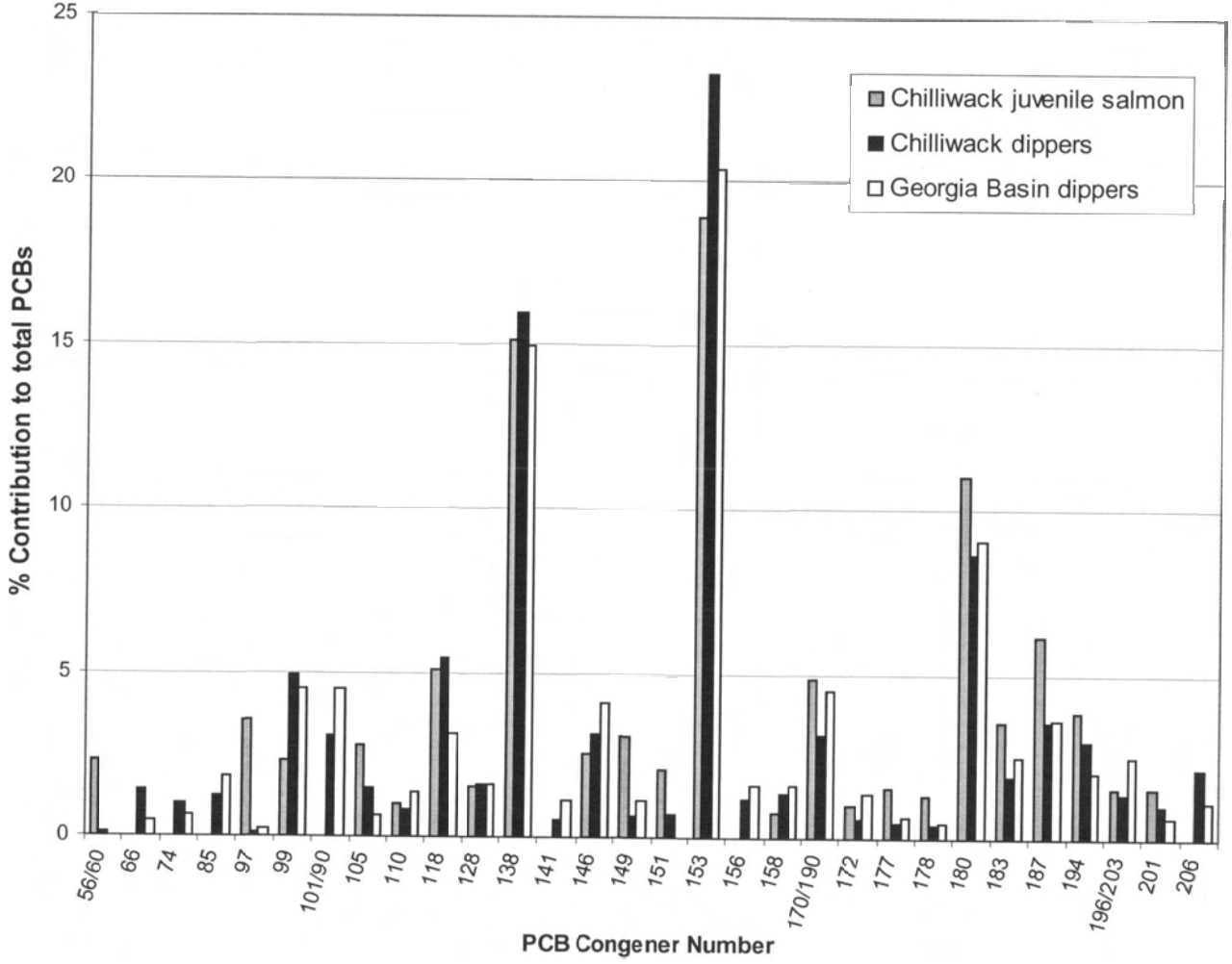
25
20
15
10
5
0
% Contribution to total PCBs
Chilliwack juvenile salmon
Chilliwack dippers
Georgia Basin dippers
56/60
66
74
85
97
99
101/90
105
110
118
128
138
141
146
149
151
153
156
158
170/190
172
177
178
180
183
187
194
196/203
201
206
PCB Congener Number

Figure 4.5

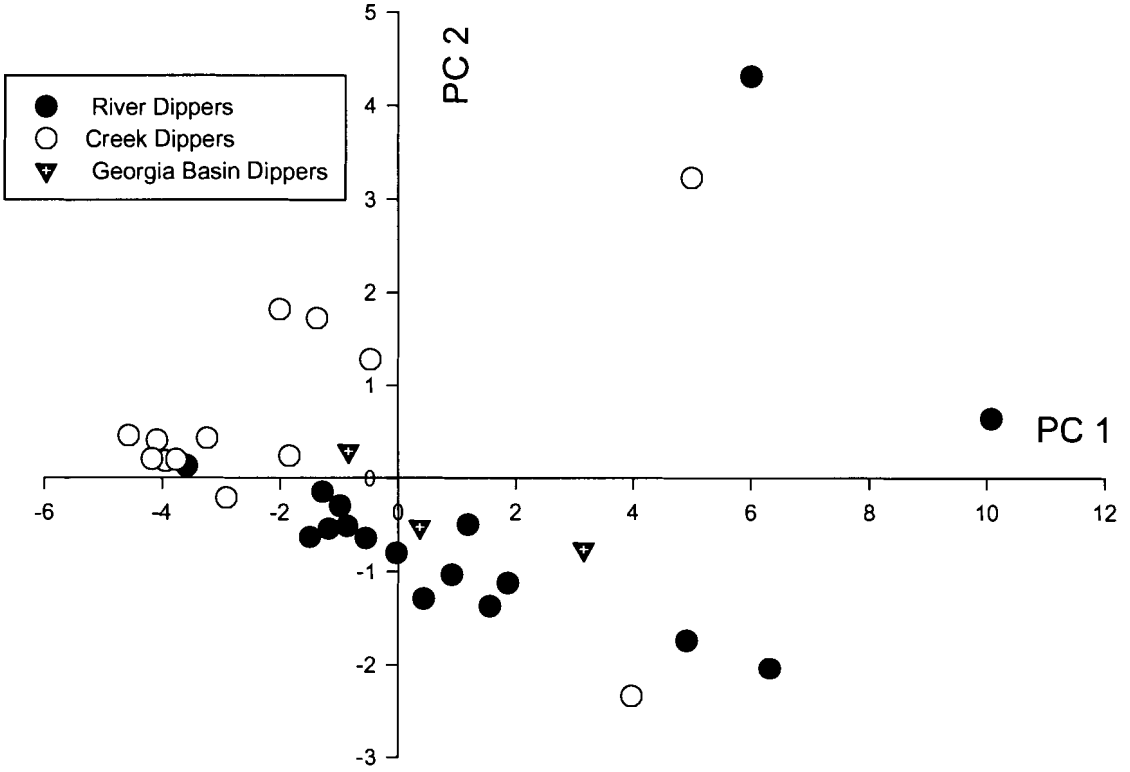
River Dippers
Creek Dippers
Georgia Basin Dippers
PC 2
PC 1
5
4
3
2
1
0
-1
-2
-3
-6
-4
-2
0
2
4
6
8
10
12

TABLE 4.1. Summary of geometric mean values (range) in ng/g wet weight for commonly detected organic contaminants in the Chilliwack watershed food chain: benthic invertebrates (creek and river locations), salmon fry, and American dipper eggs. Biomagnification factors (BMF) are calculated from dipper diet comprised of 67 % invertebrates (I) and 33 % salmon fry (F) to dipper eggs.

Sample Type	N	% Moisture	% Lipid	Total PCBs	*p,p'* DDE	HCB	*Trans*-nonachlor
Creek Invertebrates	7	80.9	2.6	**ND**[a]	**1.0** (0.8-1.4)	**0.3** (0.1-0.4)	**0.1** (0.1)
River Invertebrates	8	79.2	3.6	**ND**	**1.4** (1.3-1.8)	**0.3** (0.2-0.4)	**0.3** (0.1-0.4)
Salmon Fry	8	82.7	2.4	**0.6** (ND-30.2)	**3.3** (1.2-19.5)	**0.4** (0.2-0.6)	**0.3** (0.2-0.4)
American Dipper Eggs	32	76.2	6.4	**52.4** (3.9-706.6)	**58.5** (4.1-235.4)	**3.1** (0.9-10.1)	**0.9** (0.7-4.2)
Dipper Diet (67% I: 33% F)				**0.3**	**1.9**	**0.3**	**0.2**
BMF				**174.7 (87.3)**[b]	**30.8**	**10.3**	**4.5**

[a]ND = not detected, sample detection limit for all compounds is 0.1 ng/g wet weight.
[b]BMF value calculated from salmon fry only to dipper eggs.

TABLE 4.2. Concentrations of organic contaminants (μg/g wet weight) detected in American dipper egg samples from the Chilliwack River watershed (geometric means) and from 3 other rivers (single samples) in the Georgia Basin, British Columbia.

	Chilliwack River (n = 32)	**Coquitlam River (n = 1)**	**Cheakamus River (n = 1)**	**Seymour River (n = 1)**
% Moisture (± SD)	76.2 ± 6.9	NC[a]	80.4	86.2
% Lipid (± SD)	6.4 ± 1.4	4.2	8.1	7.3
Total PCBs	0.0524	0.0990	0.0550	0.0670
p,p' DDE	0.0585	0.0450	0.0250	0.0210
p,p' DDT	0.0002	0.0010	0.0001	0.0001
p,p' DDD	0.0002	0.0001	0.0001	0.0001
Hexachlorobenzene	0.0031	0.0001	0.0050	0.0020
α- HCH	0.0004	ND[b]	ND	ND
β- HCH	0.0001	ND	ND	ND
Heptachlor epoxide	0.0001	0.0001	ND	ND
Oxychlordane	0.0001	0.0020	0.0020	ND
Trans-nonachlor	0.0009	0.0020	0.0030	0.0010
Cis-nonachlor	0.0002	0.0001	0.0001	0.0001
Mirex	ND	ND	ND	ND
Dieldrin	ND	ND	ND	ND

[a]NC = not calculated, egg sample was too developed.
[b]ND = not detected, sample detection limit for all compounds is 0.0001 μg/g wet weight.

TABLE 4.3. Principle component analysis of 18 selected PCB congeners that contributed >1% to total PCBs detected in American dipper eggs collected during 1999-2001 from the Chilliwack River and other Georgia Basin rivers. Values shown are factor loadings for each variable (PCB congener) and the variance explained by each component is given in parentheses. See Figure 5 for principle component scores for each dipper egg.

Variable	Eigenvectors	
	PC1 (70.6 %)	PC2 (11.1%)
PCB 66	0.165	0.555
PCB 74	0.246	0.173
PCB 85	0.256	-0.009
PCB 99	0.259	-0.153
PCB 101/90	0.247	-0.106
PCB 105	0.184	0.522
PCB 118	0.241	0.234
PCB 128	0.253	-0.226
PCB 138	0.254	-0.249
PCB 146	0.263	-0.210
PCB 153	0.254	-0.259
PCB 170/190	0.257	-0.043
PCB 180	0.268	-0.021
PCB 183	0.260	-0.109
PCB 187	0.215	-0.010
PCB 194	0.243	0.231
PCB 196/203	0.225	0.042
PCB 206	0.054	0.080

TABLE 4.4. Comparison of mean concentrations for selected organic contaminants in eggs of various passerine species in British Columbia and dippers from other countries. All values are geometric means or ranges of means if locations are different. Values given are in μg/g wet weight. [a]NR = not reported.

Species	Location, Year(s)	Total PCBs	DDE	HCB	Trans-nonachlor	Reference
Tree swallow *Tachycineta bicolor*	British Columbia 1990-1991	0.180	5.05	0.009	0.002	Elliott et al., 1994
Barn swallow *Hirundo rustica*	British Columbia 1990-1991	0.088	0.893	0.003	0.007	Elliott et al., 1994
House wren *Troglodytes aedon*	British Columbia 1990-1991	0.052	4.31	0.003	0.002	Elliott et al., 1994
Mountain bluebird *Sialia currucoides*	British Columbia 1990-1991	0.044	0.56	0.004	0.012	Elliott et al., 1994
American robin *Turdus migratorius*	British Columbia 1990-1991	0.051	22.0	0.002	0.016	Elliott et al., 1994
Eurasian dipper *Cinclus cinclus*	S. Norway, 1993	0.12	0.09	0.004	NR[a]	Kallenborn et al., 1998
Eurasian dipper *Cinclus cinclus*	SW Ireland, 1990-1991	0.10	0.04	0.006	NR	O'Halloran et al., 1993
Eurasian dipper *Cinclus cinclus*	E. Scotland, 1990	0.65	0.22	0.008	NR	Ormerod and Tyler, 1992
Eurasian dipper *Cinclus cinclus*	Wales, 1990-1993	0.11-0.49	0.12	0.001	NR	Ormerod et al., 2000; Ormerod and Tyler, 1992
American dipper *Cinclus mexicanus*	British Columbia, 1999-2001	0.052	0.059	0.003	0.001	This study

CHAPTER 5:

LINKING CONTAMINANT PROFILES TO THE DIET AND BREEDING LOCATION OF AMERICAN DIPPERS USING STABLE ISOTOPE ANALYSIS

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Summary

1. We investigated the influence of diet and breeding location on individual bird contaminant profiles using a color marked population of American dippers (*Cinclus mexicanus*). The population was comprised of two distinct groups: resident dippers breeding on the main river and migrant dippers breeding on the tributaries.
2. Residue analysis of egg contents revealed a spatial trend in contaminant profiles among resident and migrant American dippers in a watershed receiving deposition of atmospheric contaminants. Total organochlorines (OC), polychlorinated biphenyls (PCB), and mercury (Hg) were all significantly higher ($p < 0.0001$, $p < 0.001$, $p < 0.0001$ respectively) in eggs from river residents compared to the creek migrants. The three most prevalent organochlorine compounds in dipper eggs, DDE, hexachlorobenzene, and trans-nonachlor, were all significantly higher in eggs from the river compared to those from the creeks.
3. We hypothesized that the observed differences in contaminant concentrations were related in part to the proportion of salmon fry (*Oncorhyncus spp.)* in the diet relative to aquatic invertebrates. Stable isotope analyses using $\delta^{13}C$ and $\delta^{15}N$ were conducted on blood and feathers of dippers and other reference species in addition to invertebrate and salmon fry prey. Linear mixing models using the ^{15}N isotope in the dippers' diet and blood revealed considerable variability in the proportion of fish in the diet of American dippers (0 to 71 %). Resident birds on

the main river had significantly higher proportions of fish in the diet (42 %) compared to the migrants on the tributaries (22 %) ($p = 0.01$).

4. The difference in diet within a single watershed explained some of the observed variation in egg contaminant profiles since blood $\delta^{15}N$ values were positively correlated with total organochlorines ($p = 0.002$) in dipper eggs indicating fish may be the primary source of contamination.
5. Dipper eggs represented local conditions on the breeding site making them useful tools for biomonitoring aquatic contaminants in watersheds. However, given the distinct difference in contaminant profiles between residents and migrant dippers and the link to diet, our results emphasize the importance of understanding the ecology of the study species for assessing toxicological effects at the population level.

Key words: biomonitor, stable isotopes, ^{15}Nitrogen, salmon fry, rivers, organochlorines, PCBs, mercury, selenium

Introduction

Aquatic ecosystems and human civilization both rely on clean sources of running water to maintain biodiversity and to supply a range of consumer processes for industrial, domestic and agricultural systems. Suitable biomonitors are therefore required for accurate assessment of impacts from local and long-range sources of pollution to streams and rivers that supply running water. The aquatic songbird, the American dipper (*Cinclus mexicanus*) occupies mountainous watersheds year-round and has proven useful as a biomonitor of metal pollution (Strom et al. 2002). The European dipper (*Cinclus cinclus*) has also been shown to be a suitable indicator of organochlorine, PCB and mercury pollution in upland catchments (Ormerod and Tyler 1990, 1992, O'Halloran et al. 1993, 2003, Ormerod et al. 2000). Previous studies have suggested that American dippers have distinct altitudinal patterns of migration, which includes seasonal movement upstream and downstream within a watershed (Bakus 1959a,b, Price and Bock 1983). Our earlier research further revealed that resident and altitudinal migrant dippers shared common wintering grounds on the river, but most migrants moved upstream onto creeks in the spring while residents remained on the river to breed (Morrissey et al. submitted). One consequence of even small-scale migratory patterns is its influence on contaminant exposure, producing large variation in observed contaminant concentrations between individuals (Hebert 1998). Furthermore, dietary composition may vary within a breeding range of individual species due to spatial segregation, which further contributes to large variability in contaminant levels (Hebert et al. 1997, Bustnes et al. 2000, Bearhop et al. 2000).

American dippers feed on benthic invertebrates and small fish during the breeding season, however, the availability of salmon fry to migrants on smaller tributaries is lower compared to that of residents along the main stem of rivers because of differences in distribution of spawning salmon (Crisp 2000). This difference in diet may be important in understanding American dipper exposure to persistent compounds given that fish tend to bioaccumulate and biomagnify persistent organic contaminants to higher concentrations than lower trophic level invertebrates (Suedel et al. 1994, Kiriluk et al. 1995, Kidd et al. 1995).

Recent use of stable isotopes in environmental and ecological studies has been a powerful means for distinguishing and tracing dietary sources. Existing applications have focused on determining trophic levels in ecosystems, separating terrestrial and marine food webs as well as looking at local differences in diets and movements of individuals (reviews by Peterson and Fry 1987, Lajtha and Michener 1994, Hobson 1999, Kelly 2000). The use of stable carbon and nitrogen isotopes can provide a means of determining diet and feeding locations and can provide valuable insight in contaminant studies (Kidd et al. 1995, Jarman et al. 1996, Hebert et al. 1997, Bearhop et al. 2000. Braune et al. 2002).

Stable isotope ratios of stream biota are well documented. Freshwater food chain studies indicate that animals feeding on fish in addition to invertebrates should have an enriched isotopic signature compared to those animals feeding almost exclusively on invertebrates (Rounick and Hicks 1985, Fry 1991, Cabana and Rasmussen 1994). Isotopic analysis of avian whole blood provides information on short-term dietary sources of assimilated foods (half life time is typically 10-16 days for ^{13}C and 9-15 days

for ^{15}N), while analysis of avian feathers give a record of past dietary information at a time when the feathers are grown (Hobson and Clark 1992, Bearhop et al. 2002, Evans Ogden 2002). By using $\delta^{15}N$ and $\delta^{13}C$ values in blood and feathers, we can infer the relative diet composition of the American dipper and further assess trophic level differences in individuals breeding at different locations within a watershed.

In this study, we used a population of individually color marked birds to determine the degree of exposure to contaminants by analyzing egg samples from known residents and migrants separately. Our first objective was to test whether the population's migratory patterns had an effect on levels of persistent compounds (organochlorines, polychlorinated biphenyls, mercury and selenium) in American dipper eggs. Additionally, since resident dippers on the river were observed eating more fish than the migrants on the creeks, we further investigated whether the diet differed between residents and migrants using stable isotope techniques. We hypothesized that each group would have a distinct isotopic signature of $\delta^{13}C$ and $\delta^{15}N$ isotopes in the blood and feathers. This would not only identify the relative contribution of fish and invertebrate sources to the diet, but could also aid in interpreting any observed differences in contaminant profiles.

Methods

STUDY AREA

The study area is located within the Chilliwack River watershed (49°1'N, 121°4'W) in the Fraser Valley, which is approximately 100 km east of Vancouver in the Cascade Mountain Range of southwestern British Columbia, Canada. Because of its unique geographic features, prevailing winds, and the large population found in the lower

reaches of the valley, the upper valley experiences interactions of urban, suburban, marine, and agricultural emissions of pollutants and their subsequent transformation into ambient air. Thus, the Chilliwack watershed receives atmospheric inputs of organic and inorganic pollutants through wet and dry deposition processes (Morrissey 2003). Annual precipitation averages 1850.5 mm with mean daily temperatures of 10.4°C (Data from 1879-1990). The watershed tributaries are first through third order streams. The river is dominantly a fourth order stream, fed by a large glacial lake at its upper end and is 43.5 km in length where it merges to become the Vedder River. The Chilliwack River supports populations of Pacific chum, coho, pink, and chinook salmon (*Oncorhynchus keta, O. kisutch, O. gorbuscha, O. tshawytscha*) as well as cutthroat trout (*O.clarki*), steelhead trout (*O. mykiss*), and Dolly Varden (*Salvelinus malma*) (B.C. Fisheries data). Annual breeding runs of anadromous salmon and steelhead spawn from late summer through winter within the watershed, but peak runs occur along the main stem of the river and at the hatchery in autumn. Salmon fry are therefore most abundant in early spring, which coincides with the breeding period of the American dipper.

SAMPLING PREY

Fifteen composite samples of benthic invertebrates were collected at eight different sites along the main stem of the river and from seven different tributaries in the watershed in April 2001. Aquatic larval invertebrates (~ 1 g dry weight) were collected either by kick sampling in the stream (disturbing the rocks directly upstream of a Surber sampler) or by turning over rocks by hand. Each sample represented a mixture of macro-invertebrates primarily comprised of Ephemeropteran, Plecopteran and Tricopteran larvae that dippers are known to naturally prey upon (Mitchell 1968, Ealey 1977).

During invertebrate collections, eight additional samples comprised of approximately ten salmon fry (*Oncorhynchus spp.)* (age-0), were collected live using a dip net from the same eight sites along the main river. This represented a composite sample of predominantly coho and chum salmon fry (~80 %) but pink and chinook salmon (~20 %) were also included. Higher water flows and lower densities of salmon fry on tributaries prevented us from capturing fish on creeks to include in the analysis. Both the invertebrates and fish were held alive in stream water in polyethylene bags for several hours to allow the stomach contents to purge to obtain accurate isotopic values of the tissues. All samples were subsequently washed three times with distilled deionized water to remove any surface contamination or stream water and stored frozen in glass vials until preparation for stable isotope ratio analysis.

SAMPLING AMERICAN DIPPERS AND REFERENCE SPECIES

Adult American dippers were caught throughout the breeding season from late March until July in all areas of the watershed both on the main river and on the associated tributaries. Dippers were captured using 6m passerine mist nets set up over moving water in narrow channels or on edges of the river and creeks. A hand-net was used to trap a few additional birds while on the nest during incubation (females) or during nest building (males). All dippers were weighed, measured and banded with a numbered USFWS aluminum band and three colored celluloid bands to represent a unique combination that was readily identifiable when the birds were resighted. We drew approximately 0.7 - 0.8 ml of blood from the jugular vein using a syringe and needle for most individuals (Kerlin 1964, Hoysak and Weatherhead 1991) or 0.1 - 0.2 ml in a few individuals by puncturing the metatarsal vein and collecting blood in capillary tubes. In

addition, 10 - 15 breast feathers were removed and stored in polyethylene bags prior to the bird's release. We randomly selected 31 individual blood samples (16 from the river, 15 from the creeks) and 20 individual feather samples (10 from the river, 10 from the creeks) for analysis of stable isotope ratios. Blood and feather samples used for the stable isotope analysis were not always from the same individuals because we did not obtain blood samples from all birds and used many feather samples for other analyses unrelated to the present study.

In addition to the American dippers, we collected blood and feathers from two reference groups in the Chilliwack River watershed. Common mergansers (*Mergus merganser*) were considered a model for an obligate piscivore and swallows (3 species: *Tachycineta thalassina, Tachycineta bicolor, Stelgidopteryx serripennis*) were considered a model for an obligate insectivore. Common mergansers (n = 12) were captured using waterfowl mist-nets set up across sections of the river and swallows (n = 9) were captured in the nest cavity using a hand-net over the opening. During mist netting for mergansers, we opportunistically captured a single Belted Kingfisher (*Ceryle alcyon*), an obligate piscivore, from which we also sampled blood and feathers to be added to the merganser samples. All individuals were banded with a USFWS metal band, which prevented duplicate sampling of the same individual. We collected 1.0 ml of blood from the brachial vein of mergansers with a syringe and needle and 0.1 ml of blood from the brachial vein of the swallows and kingfisher by puncturing the vein and drawing off blood with capillary tubes. In addition, several breast feathers were taken from each individual and stored in sealed polyethylene bags. All sample collections were done in accordance with Animal Care committee approval and with a valid Environment Canada

scientific permit. We observed no negative effects from blood or feather sampling of individuals. Furthermore, egg collections and handling of breeding birds was never observed to cause nest abandonment or hatching failure.

EGG COLLECTION AND CONTAMINANT RESIDUE ANALYSIS

A single egg from a clutch of four or five eggs was collected from dipper nests throughout the watershed in 1999, 2000 and 2001. Viable and non-viable eggs were selected at random from each of 33 different clutches from resident dippers on the main river (n= 17) and from migrants on the tributaries (n = 16). Considerable variation in contaminant levels has been observed within and between clutches allowing for equal treatment of eggs regardless of their position in the clutch or in repeat clutches (Morrissey unpubl. data). Whole eggs were stored refrigerated for up to four weeks. Egg contents were then transferred into an acetone: hexane rinsed glass jar to be frozen at – 25°C until analysis.

Chemical analyses of all egg samples were carried out at the National Wildlife Research Centre (NWRC) Hull, Quebec. Organochlorine analyses included determination of chlorobenzenes (tetrachlorobenzene, pentachlorobenzene, hexachlorobenzes), hexachlorocyclohexanes (α-,β-,γ-HCH), chlordane related compounds (oxychlordane, trans-chlordane, cis-chlordane, trans-nonachlor, cis-nonachlor and heptachlor epoxide), DDT and metabolites (p,p'-DDE and p,p'-DDD), mirex and photomirex, and dieldrin. Total PCBs were calculated by summing the peaks of 62 individual congeners identified. Samples were quantitatively analyzed by capillary gas chromatography coupled with a mass selective detector operated in selected ion monitoring mode according to CWS Method No. MET-CHEM-OC-04B (Won et al.

2000). Briefly, samples underwent neutral extraction with 1:1 DCM/Hexane after dehydration with anhydrous sodium sulfate, removal of lipids and biogenic materials by Gel Permeation Chromatography, and further cleanup by Florisil column chromatography. All samples to be analyzed were spiked with internally labeled ^{13}C standards prior to extraction. Each sample extract was injected twice, once for determination of organochlorines and once for PCBs. As part of the quality control, blanks and CWS reference material (1989 Lake Ontario Herring Gull QA) were run concurrently with the samples. The nominal detection limit for all compounds was 0.0001 μg/g wet weight. Internal standard recoveries were typically between 80 % and 110 % and residues were not recovery corrected. Egg concentrations are reported on a wet weight basis (Peakall and Gilman 1979) with arithmetic corrections of desiccated samples that deviated by more than 5 % from the mean moisture content (78.6 %) of freshly collected, undeveloped eggs.

Total mercury (Hg) and selenium (Se) were analyzed according to CWS Method No. MET-CHEM-AA-02E (Neugebauer et al. 2000). The sample homogenates were freeze-dried to determine moisture content and analyzed for mercury without prior acid digestion on the AMA-254, which employs direct combustion of sample in an oxygen rich atmosphere. Samples for which there was sufficient material remaining were digested in nitric acid according to standard techniques for selenium analysis. Selenium was analyzed by graphite furnace atomic absorption spectrometry (GFAAS) using a Perkin Elmer 3030b equipped with a Deuterium background corrector and HGA-300 Graphite furnace. Accuracy of analysis was determined using certified reference materials Dolt-2 and Dorm-2 (National Research Council of Canada) and blank samples.

Recoveries of reference materials were within certified range (95 % to 121 %). Additionally, random egg samples were analyzed in duplicate to check precision. All values for mercury and selenium are reported on a dry weight basis and detection limits for under these conditions were 0.18 μg/g dry weight for mercury and 0.10 μg/g dry weight for selenium.

STABLE ISOTOPE ANALYSIS

Invertebrate and fish samples were stored frozen and then freeze-dried for 24 to 48 hours until completely dry and then ground up to a fine powder. Three separate sub-samples of each of the composites of invertebrates and fish were analyzed to ensure homogeneity of the mixture, particularly the invertebrate taxa. Whole blood samples were also stored frozen until preparation for determination of stable isotope ratios. Samples were then freeze-dried for 24 hours and homogenized. Feather samples were washed with a 2:1 chloroform and methane solution and thoroughly rinsed with distilled deionized water to remove any surface lipids or external contamination. They were subsequently oven dried and cut-up into tiny pieces. A sub-sample of each tissue (~1 mg) was weighed into miniature tin capsules (5 x 9 mm, Costech Analytical) for combustion at over 1000°C using an online elemental analyzer.

Stable isotope ratios of carbon and nitrogen were analyzed at the Stable Isotope Facility, University of Davis, California using a Europa 20/20 continuous flow isotope ratio mass spectrometer (CFIRMS). Sample isotope ratios were compared to the standard gases Pee Dee Belemnite for ^{13}C, and atmospheric nitrogen (AIR) for ^{15}N that were injected directly into the CFIRMS before and after the sample peaks. Values for ^{13}C and

^{15}N were calculated and reported using the standard delta (δ) notation in parts per thousand (‰) as follows:

$\delta X = [(R_{sample}/R_{standard}) - 1] \times 1000$,

where X is ^{15}N or ^{13}C and R is the corresponding ratio $^{15}N/^{14}N$ or $^{13}C/^{12}C$. Replicate laboratory standards (ammonium sulfate ^{15}N = 1.33 ‰ and sucrose ^{13}C = -23.83) were analyzed before and after every 12 samples to determine the accuracy of ^{13}C and ^{15}N values. Measurement errors averaged ± 0.1 ‰ for nitrogen, and ● 0.04 ‰ for carbon.

DATA ANALYSIS

A two-sample t-test was used to compare mean total organochlorines ($\sum$OCs), total polychlorinated biphenyls ($\sum$PCBs), mercury (Hg), and selenium (Se) residues in resident and migrant American dipper eggs. Organochlorine and total PCB residues were not normally distributed (Shapiro-Wilk W test); therefore values were log transformed to obtain normal distributions prior to performing parametric statistical tests. Values are reported as arithmetic means ● SE and geometric means for OCs and PCBs. Where levels were below the detection limit for organochlorine compounds in eggs, a value equal to the minimum detection limit (0.0001 μg/g wet weight) was applied. For mercury, we used a value of one half the detection limit (0.09 μg/g) to permit statistical analyses. Sample size for OC/PCB analysis was 32 eggs, 33 for mercury and 27 for selenium based on available sample mass for each analysis.

We determined differences in stable isotopic signatures ($\delta^{15}N$ and $\delta^{13}C$) among prey types (creek invertebrates, river invertebrates and salmon fry), using a K- nearest neighbor randomization test (Rosing et al. 1998), which treats the isotopic signatures of $\delta^{13}C$ and $\delta^{15}N$ as spatial data. The same analysis was then used to determine differences

in blood and feather isotope values among resident and migrant dippers and between reference species (mergansers and swallows). Where appropriate, an ANOVA was used to detect differences in mean $\delta^{15}N$ values among groups followed by a multiple comparison Tukey HSD test. Statistical analyses were performed using JMP v.4.0 (SAS Institute).

A relative index of the invertebrate and fish component of the diet was calculated for each dipper as well as a mean for each group using only the $\delta^{15}N$ values in a two source, linear mixing model (Phillips and Gregg 2001). Since larger changes in $\delta^{15}N$ occur with trophic level this isotope is more sensitive to diet effects. River birds were analyzed separately from the creek birds because invertebrate $\delta^{15}N$ signatures were significantly higher on the river likely as a result of enriched marine derived nutrients from high densities of spawning salmon on the main river (Bilby et al. 1996, Johnston et al.1997). Isotope ratios of salmon fry collected from the river were assumed to be representative of both river and creek locations since we were unable to capture fish on tributaries due to their lower frequency and high water flows. We used fractionation values of 2.9 ‰ for $\delta^{15}N$ in invertebrate samples and 1‰ for $\delta^{15}N$ in fish samples that were calculated from the two reference species (swallows and mergansers/kingfisher). These were comparable to values obtained in captive feeding experiments of birds on high protein diets (Evans Ogden 2002, Bearhop et al. 2002). Proportions of fish in the diet were transformed using an arcsine transformation to permit parametric statistical comparisons. Male and female $\delta^{15}N$ values and relative proportions of fish in the diet were compared using a two-sample t-test to determine if sex biases existed in diet

composition. Combined sex isotope data were then used to represent overall diet composition for comparison to contaminant profiles in dipper eggs.

Results

CONTAMINANT RESIDUES IN AMERICAN DIPPER EGGS

Mean concentrations of total organochlorines ($t_{29} = -4.9$, $p < 0.0001$) and total polychlorinated biphenyls ($t_{28} = -3.6$, $p = 0.001$) were higher in eggs from river residents compared to the creek migrants (Fig. 1). Of the organochlorine compounds, residues of DDT metabolites, chlordane compounds, and chlorobenzenes were all significantly higher in resident eggs compared to migrants (Table 1). Detection of hexachlorocyclohexane (HCHs) was low and therefore, differences between residents and migrants were marginally insignificant. Mirex, photomirex, and dieldrin were not detected in any of the egg samples. Mercury concentrations were also found to be significantly higher in eggs of river residents compared to the creek migrants ($t_{31} = -4.7$, $p < 0.0001$) (Table 2). However, there was no difference in selenium concentrations between the migratory groups ($t_{25} = -1.15$, $p = 0.26$).

STABLE ISOTOPE ANALYSIS

Analyses of prey samples from the main river channel and the tributaries indicated significant differences in isotopic signatures between river invertebrates, creek invertebrates and fish (K nearest neighbour randomization test, $p < 0.0003$) (Table 3). The $\delta^{15}N$ values in creek invertebrates were significantly different from river invertebrates ($F = 239.2$, $p<0.0001$), therefore, we proceeded to test for trophic level differences separately for resident dippers on the main stem of the river and for migrant dippers on the tributaries.

Differences in isotopic signatures were detected in whole blood samples collected from the reference species (swallows and mergansers) and American dippers (river resident and creek migrant dippers) (Figure 2). The K nearest neighbor randomization test indicated all four groups were significantly spatially separated ($p < 0.0001$). $\delta^{15}N$ values were also isotopically different among groups ($F_{3,48} = 79.8$, $p < 0.0001$) with no overlap of confidence intervals. River resident dippers had a $\delta^{13}C$ and $\delta^{15}N$ signature that was intermediate between the two reference species; while creek dippers could not be compared because of a lack of suitable references (i.e. swallows breeding on creeks). Furthermore, river and creek dippers had significantly different blood isotopic signatures with river residents being more enriched in $\delta^{15}N$ and $\delta^{13}C$ (K nearest neighbor randomization test, $p< 0.0001$) (Fig. 3).

Contributions of fish and aquatic invertebrates to the diets of the American dipper during the breeding season were analyzed using a two source linear mixing model with a single isotope ^{15}N. Since larger changes in $\delta^{15}N$ occur with trophic level, this isotope is more sensitive to diet effects than $\delta^{13}C$. Additionally, we found considerably less variability among prey samples for $\delta^{15}N$ allowing for increased precision in estimating diet composition. The model provided an index of the proportional consumption and assimilation of the diet sources. The amount of fish in the diets of American dippers ranged widely from 0 % to 71 % (Fig.4). Male and female dippers showed no significant differences in $\delta^{15}N$ values ($t_{29} = 0.20$, $p = 0.8$), and thus no differences in diet composition ($t_{29} = 0.96$, $p = 0.4$) (Table 4). However, resident dippers on the river had a higher mean percentage of fish in the diets (42 % ± 7 %) compared to migrants on the tributaries (22 % ● 6 %) ($t_{29} = -2.7$, $p = 0.01$).

Patterns of isotopic signatures in feather samples from the reference species and the dippers were not consistent with blood samples. There was a lack of observable patterns in feather isotopic signatures among species with respect to diet. Feathers of the mergansers and swallows were significantly spatially separated (K nearest neighbor randomization test, $p = 0.003$) but with considerable variation among individuals. Spatial separation of isotopic signatures between resident and migrant dipper feathers was marginally insignificant (K nearest neighbor randomization test, $p = 0.06$) (Fig.5). Given the lack of any clear distinction between resident and migrant dippers, or suitable end points from the reference species, we could not assess the relative contributions of fish and aquatic invertebrates to the diets of dippers during the period of feather growth in the previous year. Therefore, we could not definitively infer molting location by feather isotope signatures.

Although we did not set out to collect blood and eggs from the same individuals within a territory to compare isotope ratios with contaminant residues, we had a small sample ($n = 8$) for which we had data on both egg contaminants and blood stable isotope ratios collected from a bird in the same territory in the same season (4 river residents and 4 creek migrants). Residues of log wet weight total organochlorines in dipper egg samples were positively correlated with $\delta^{15}N$ values in blood from birds sampled from the same territory ($r = 0.91$, $p = 0.002$). Given that DDT and metabolites made up over 90 % of the organochlorines pesticide residues in dipper eggs, the log sum of DDT metabolites was also significantly correlated with $\delta^{15}N$ values in blood ($r= 0.91$, $p < 0.002$) (Fig. 6a). Correlation between total PCBs and $\delta^{15}N$ values in dipper blood was marginally insignificant ($r = 0.68$, $p = 0.06$) (Fig. 6b). Levels of total organochlorines

were not intercorrelated with total PCBs ($p = 0.3$). There was no relationship between egg mercury levels and $\delta^{15}N$ values in blood, primarily because many samples from creeks were below detection limits for mercury.

Discussion

SOURCES AND SIGNIFICANCE OF CONTAMINANT RESIDUES

Relatively few studies have noted the importance of small-scale migratory movements on the interpretation of contaminant concentrations in focal species. Studies that do report such movements have shown they are a significant factor in explaining individual variability in contaminant concentrations. Hebert (1998) noted that winter weather influenced migratory movements of Herring gulls in the Great Lakes, which ultimately had an effect on the birds' exposure to local organochlorine contamination. Our data on the American dipper further demonstrate the importance of considering spatial trends in species that are classified as non-migratory.

Variability in contaminant concentrations in American dipper eggs was largely explained by the bird's status as a resident on the main river or an altitudinal migrant on one of the watershed tributaries. Each group had significantly different levels of organochlorines, PCBs and mercury in their eggs, with resident eggs showing higher concentrations of all contaminants measured, except selenium. This trend was apparent despite the fact that the system is a single continuous watershed where distances separating the groups are relatively minor (<1 to 15 km). In other studies, we found that the residents and migrants wintered on the main river together, but the migrants moved upstream onto the tributaries to breed (Morrissey et al. submitted). Our data implies that the degree of contamination in eggs is largely determined by exposure on the breeding

site, and that nutrients are deposited in the eggs primarily from recent dietary uptake. This finding is consistent with those by Ormerod et al. (2000), who suggested that eggs of the European dipper also reflected local sources of contamination. Most nutrients required for egg formation in passerines are not stored prior to laying but are gathered on a daily basis during the laying period (Perrins 1996). This is not unexpected given that egg formation in passerines is costly in terms of energy and nutrient requirements where the combined mass of the eggs in the clutch can often equal or outweigh the female's own body weight (Perrins 1996). Therefore, recent dietary intake in dippers will largely influence contaminant levels found in eggs making them a useful biomonitor of local contamination.

Given that persistent lipophilic organic compounds are known to bioaccumulate and biomagnify with increasing trophic levels (Connel 1990), dippers with a larger proportion of fish in their diet were expected to have higher exposure to chlorinated hydrocarbons as compared to those primarily on an invertebrate diet. The results from the stable isotope data support this hypothesis, demonstrating that river residents were consuming a larger proportion of fish during the breeding season. In addition, the $\delta^{15}N$ signature in blood of banded birds was significantly correlated with the log of $\sum$DDT and the log of total organochlorines in eggs despite the limitations of sample size. These results were consistent with other studies where an increased ^{15}N signature was associated with a higher trophic level and subsequently higher contaminant burdens (Broman et al. 1992, Kidd et al. 1995, Jarman et al. 1996, Bearhop et al. 2000).

Differences in location between river residents and creek migrants within the watershed may have further contributed to the observed contaminant trends. Patterns

have been shown to exist in watersheds with respect to stream ecosystem structure and function (e.g. the River Continuum Concept) that suggest a predictable transition from the headwaters to higher order reaches of rivers (Vannote et al. 1980). Conventional changes occur in stream flow, chemistry, nutrient levels, species richness and biomass along with increasing contaminant loads through atmospheric deposition from upstream to downstream (Giller and Malmqvist 2000). This effect can explain some of the variability in contaminant profiles between resident and migrant dippers since residents breed on the higher order river and migrants breed on low order tributaries. However, we have found that levels of some contaminants such as mercury and DDE did not differ significantly in benthic invertebrate prey collected from river and creek locations (Morrissey 2003). Instead, mercury and DDE were detected at a higher frequency and at higher concentrations in fish tissues compared to invertebrates. Additionally, PCBs were not detected in any invertebrate samples regardless of location, but were detected in salmon fry samples indicating fish are the primary source of exposure (Morrissey 2003). Therefore, the influence of diet may be more significant than breeding location for several contaminants of concern.

With the exception of selenium which was at or near a threshold level of 3 μg/g dry weight for reproductive toxicity (Lemly 1993), the levels of organochlorines, PCBs and mercury in dipper eggs were all relatively low and generally below levels known to cause toxicity (Beyer et al. 1996). Despite the low concentrations, many contaminants were detectable with a highly significant trend existing among residents and migrants. This indicates that American dipper eggs are useful tools for monitoring organic pollutants in watersheds. American dipper blood samples have also been successfully

used to indicate lead pollution in a mine-impacted stream (Strom et al. 2002). Future studies using dippers as monitors of water quality in impacted watersheds are expected to emerge as anthropogenic developments threaten streams in western North America. However, consideration as to the differences in location and ecology of residents and migrants will be an important factor in interpreting the source of contaminant burdens in dippers.

BLOOD AND FEATHER ISOTOPE ANALYSIS

Based on controlled laboratory studies of captive birds, whole blood has been shown to be a useful tissue for stable isotope analysis since it incorporates both ingested and assimilated diet items over a window of 20 to 30 days (Hobson and Clark 1992, Bearhop et al. 2002). In passerines, this information can be directly used to identify the diet and nutrient sources at the time of egg formation but will depend on the timing of blood sampling and the consistency of diet during that time window.

The difference in blood isotopic signature between residents and migrants was related to the proportion of fish in the diet but also to the relative isotopic signature from the aquatic invertebrates. Since marine derived nutrients are enriched in ^{15}N, spawning salmon contribute their marine signature to the stream benthos through carcass decay (Bilby et al. 1996, 1998, Johnston et al.1997). Salmon abundance is considerably higher on the Chilliwack River relative to the tributaries (B.C. Fisheries Data 1985-2000). Many salmon spawn in high densities along rivers but fewer fish migrate upstream onto the tributaries because of steep gradients, higher flows and the presence of barriers (Crisp 2000). Therefore, ^{15}N signatures in benthic invertebrates on the creeks were considerably lower than river invertebrates. We assumed isotope ratios of salmon fry collected from

the river were representative of both river and creek locations since we were unable to capture fish on tributaries due to their lower frequency and high water flows. If salmon fry are significantly depleted in ^{15}N on the tributaries as a result of lower spawning densities, we may have underestimated the proportion of fish in the diet of creek birds. However, differences in stable isotope values of salmon fry caused by lower salmon densities on the creeks are likely less than 10-20 % for $\delta^{15}N$ and only 2 % for $\delta^{13}C$ (Johnston et al.1997, Bilby et al. 1998). This equates to a relative increase of 3-6 % in the mean proportion of fish in creek bird's diet (max. 28 % fish instead of 22 %). Linear mixing models have considerably greater error associated with them than this (Ben-David and Schell 2001); therefore our estimates for diet proportions should be taken as an index rather than an absolute value of diet proportions. Ultimately, isotopic signatures were generally higher in river dippers compared to creek birds because of the higher consumption of salmon fry by river residents during the breeding season, and the higher densities of spawning salmon, which contribute marine derived nitrogen to the river biota.

The isotopic signature in feathers reflects the diet during the period of regrowth and isotopic compositions are essentially locked into the feather structure post-molt (Hobson and Clark 1992, Mizutani et al. 1992). It was apparent that the reference species did not have an isotopic signature in the feathers that resembled that of their blood samples. We have no information on where the mergansers and swallows grew the feathers; therefore, we could not use them to determine isotopic values for obligate fish and insect feeders to compare with American dippers.

Since dippers tend to molt when the breeding season is complete (usually from July to September) (Kingery 1996), stable isotope analysis can give information on the previous year's breeding and molting location based on known isotopic signatures of residents and migrants. This would allow us to use feathers as simple tools to identify the breeding location of a bird even in winter when birds are more easily caught. Although resident and migrant dippers showed some similarities between feathers and blood with respect to diet, clearly not all the birds had the same pattern. Mizutani et al. (1992) showed that both ^{13}C and ^{15}N were enriched in feathers relative to the diet of nine species of aquatic birds. Therefore, we would expect that if resident and migrant dippers were molting in the same location as they bred in the preceding year, their feather isotopic signatures should show a similar pattern as the blood values. Notably, at least two individual residents and two migrants had an alternate isotopic signature that did not reflect that of the majority. Since these birds were color banded and we knew their previous year's breeding location, these results were not expected. A change in diet during the period of feather growth may have caused this result or a movement away from the breeding site for molting. We assumed at the outset of the study that dippers were breeding and molting in the same location given that individuals we have sighted during the molt had not changed locations. However, some dippers were not seen on their territories for days or weeks following breeding. Price and Bock (1983) similarly reported that American dippers in his study area of Colorado were frequently off their territories while molting or were not visible during that period. Dippers are highly secretive and often flightless during the molt cycle; therefore, some individuals may seek out refuge in new locations or change their diet, ultimately altering their isotopic

signature. Therefore, further research is needed to determine if feather isotope analysis is a useful technique for identifying breeding and molting locations in this species.

SYNTHESIS AND APPLICATIONS

Given that the degree of contamination in dipper eggs is largely determined by exposure on the breeding site, and that nutrients are deposited in the eggs primarily from recent dietary uptake, dipper eggs can be used as an effective tool for monitoring local contaminant conditions. Although there appeared to be no additional effect of migration on egg contaminant profiles, spatial segregation as a result of differential migration strategies, proved an important factor in exposure to organic contaminants. As such, contaminant levels in dipper eggs reflected prey selection of the individual. Therefore, if we are to correctly assess trends in contaminant concentrations inferred by indicator species such as the dipper, we must first have a sound understanding of the population's structure, migration strategy and diet. We recommend that results of future contaminant studies on this species and other indicator species be interpreted with a more integrative approach, which accounts for the spatial variation in breeding sites and relative prey availability.

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References

Bakus, G.J. (1959a) Observations on the life history of the Dipper in Montana. *Auk*, **76**, 190-207.

Bakus, G.J. (1959b) Territoriality, movements and population density of the Dipper in Montana. *Condor,* **61,** 410-425.

Bearhop, S., Waldron, S., Thompson, D. & Furness, R. (2000) Bioamplification of mercury in the Great Skua *Catharacta skua* chicks: the influence of trophic status as determined by stable isotope signatures of blood and feathers. *Marine Pollution Bulletin*, **40**, 181-185.

Bearhop, S., Waldron, S., Votier, S.C. & Furness, R.W. (2002) Factors that influence assimilation rates and fractionation of nitrogen and carbon stable isotopes in avian blood and feathers. *Physiological and Biochemical Zoology*, **75**, 451-458.

Ben-David, M. & Schell, D.M. (2001) Mixing models in analyses of diet using multiple stable isotopes: a response. *Oecologia*, **127**, 180-184.

Beyer, W.N., Heinz, G.H. & Redmon-Norwood, A.W. (eds.) (1996) *Environmental contaminants in wildlife: interpreting tissue concentrations*. CRC Press, Boca Raton, Florida, 494 pp.

Bilby, R.E., Franson, B.R. & Bisson, P.A. (1996) Incorporation of nitrogen and carbon from spawning coho salmon into the trophic system of small streams: evidence from stable isotopes. *Canadian Journal of Fisheries and Aquatic Science*, **53**, 164-173.

Bilby, R.E., Franson, B.R., Bisson, P.A. & Walter, J.K. (1998) Response of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead (*Oncorhynchus mykiss*) to the addition of salmon carcasses to two streams in southwestern Washington, U.S.A. *Canadian Journal of Fisheries and Aquatic Science*, **55**, 1909-1918.

Braune, B.M., Donaldson, G.M. & Hobson, K.A. (2002) Contaminant residues in seabird eggs from the Canadian Arctic. II. Spatial trends and evidence from stable isotopes for intercolony differences. *Environmental Pollution*, **117**, 133-145.

Broman, D., Näf, C., Rolff, C., Zebühr, Y., Fry, B. & Hobbie, J. (1992) Using ratios of stable nitrogen isotopes to estimate bioaccumulation and flux of polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in two food chains from the Northern Baltic. *Environmental Toxicology and Chemistry*, **11**, 331-345.

Bustnes, J.O., Erikstad, K.E., Bakken, V., Mehlum, F. & Skaare, J.U. (2000) Feeding ecology and the concentration of organochlorines in Glaucous gulls. *Ecotoxicology*, **9**, 179-186.

Cabana, G. & Rasmussen, J.B. (1994) Modelling food chain structure and contaminant bioaccumulation using stable nitrogen isotopes. *Nature*, **372**, 255-257.

Connel, D.W. (1990) *Bioaccumulation of xenobiotic compounds*. CRC Press, Boca Raton, Florida. 187-205.

Crisp, D.T. (2000) *Trout and salmon: ecology, conservation, and rehabilitation*. Fishing News Books, Blackwell Science, Oxford.

Ealey, D.M. (1977) *Aspects of the ecology and behaviour of a breeding population of Dippers (Cinclus mexicanus: Passeriformes) in southern Alberta*. MSc. Thesis. University of Alberta, Edmonton.

Evans Ogden, L. (2002) *Non-breeding shorebirds in a coastal agricultural landscape: winter habitat use and dietary sources*. PhD dissertation. Simon Fraser University, Burnaby, British Columbia.

Fry, B. (1991) Stable isotope diagrams of freshwater food webs. *Ecology*, **72**, 2293-2297.

Giller, P.S. & Malmqvist, B. (2000) *The Biology of Streams and Rivers*. Oxford University Press, New York.

Hebert, C.E., Shutt, J.L. & Norstrom, R.J. (1997) Dietary changes cause temporal fluctuations in polychlorinated biphenyl levels in Herring gull eggs from Lake Ontario. *Environmental Science and Technology*, **31**, 1012-1017.

Hebert, C.E. (1998) Winter severity affects migration and contaminant accumulation in Northern Great Lakes Herring Gulls. *Ecological Applications*, **8**, 669-679.

Hobson, K.A. (1999) Tracing origins and migration of wildlife using stable isotopes: a review. *Oecologia*, **120**, 314-326.

Hobson, K.A. & Clark, R.G. (1992) Assessing avian diets using stable isotopes I: turnover of 13C in tissues. *Condor*, **94**, 181-188.

Hoysak, D.J. and P. J. Weatherhead (1991) Sampling blood from birds: a technique and an assessment of its effect. *Condor*, **93**, 746-752.

Jarman, W.M., Hobson, K.A., Sydeman, W.J., Bacon, C.E. & McLaren, E.B. (1996) Influence of trophic position and feeding location on contaminant levels in the Gulf of the Farallones food web revealed by stable isotope analysis. *Environmental Science Technology*, **30**, 654-660.

Johnston, N.T., Macdonald, J.S., Hall, K.J. & Tschaplinski, P.J. (1997) A preliminary study of the role of sockeye salmon (*Oncorhynchus nerka*) carcasses as carbon and nitrogen sources for benthic insects and fishes in the "Early Stuart" stock spawning streams, 1050 km from the ocean. *B.C. Ministry of the Environment, Lands and Parks. Fisheries Project Report No. RD55*, 24 pp.

Kelly, J. F. (2000) Stable isotopes of carbon and nitrogen in the study of avian and mammalian trophic ecology. *Canadian Journal of Zoology*, **78**, 1-27.

Kerlin, R.E. (1964) Venipuncture of small birds. *Journal of American Veterinary Medical Association*, **144**, 870-874.

Kidd, K.A., Schindler, D.W., Hesslein, R.H. & Muir, D.C.G. (1995) Correlation between stable nitrogen isotope ratios and concentrations of organochlorines in biota from a freshwater food web. *Science of the Total Environment*, **160/161**, 381-390.

Kingery, H.E. (1996) American Dipper (*Cinclus mexicanus*). *The Birds of North America,* no.229 (eds. A. Poole and F. Gill), pp.1-28. The Academy of Natural Sciences, Philadelphia, and American Ornithologists' Union, Washington, D.C.

Kiriluk, R.M., Servos, M.R., Whittle, D.M., Cabana, G. & Rasmussen, J.B. (1995) Using ratios of stable nitrogen and carbon isotopes to characterize the biomagnification of DDE,

mirex, and PCB in a Lake Ontario pelagic food web. *Canadian Journal Fisheries and Aquatic Science*, **52**, 2660-2674.

Lajtha, K. & Michener, R.H.(eds.) (1994) *Stable isotopes in ecology and environmental science*. Blackwell Scientific Publications, Oxford.

Lemly, A.D. (1993) Guidelines for evaluating selenium data from aquatic monitoring and assessment studies. *Environmental Monitoring and Assessment*, **28**, 83-100.

Mitchell, P.A. (1968) *The food of the Dipper (Cinclus mexicanus Swainson) on two western Montana streams.* MSc thesis. University of Montana, Missoula.

Mizutani, H., Michio, F. & Kabaya, Y. (1992) 13C and 15N enrichment factors of feathers of 11 species of adult birds. *Ecology*, **73,** 1391-1395.

Morrissey, C.A. (2003) *The ecological and toxicological significance of altitudinal migration by the American dipper (Cinclus mexicanus).* PhD dissertation. Simon Fraser University, Burnaby, British Columbia.

Morrissey, C.A., Bendell-Young, L.I. & Elliott, J.E. (submitted) Seasonal trends in population density, distribution and movement of American dippers within a watershed of southwestern British Columbia, Canada. *Condor.*

Neugebauer, E.A., Sans Cartier, G.L. & Wakeford, B.J. (2000) Methods for the determination of metals in wildlife tissues using various atomic absorption spectrophotometry techniques. Technical Report Series No. 337. Canadian Wildlife Service Headquarters, Hull, Quebec, Canada.

O'Halloran, Ormerod, S.J., Smiddy, P., & O'Mahony, B. (1993) Organochlorine and mercury content of dipper eggs in South-west Ireland. *Biology and Environment*, **93B**, 25-31.

O'Halloran, J., Irwin, S., Harrison, S., Smiddy, P. & O'Mahony, B. (2003) Mercury and organochlorine content of Dipper *Cinclus cinclus* eggs in south-west Ireland: trends during 1990-1999. *Environmental Pollution*, **123**, 85-93.

Ormerod, S.J. & Tyler, S.J. (1990) Environmental pollutants in the eggs of Welsh Dippers *Cinclus cinclus*: a potential monitor of organochlorine and mercury contamination in upland rivers. *Bird Study*, **37**, 171-176.

Ormerod, S.J., & Tyler, S.J. (1992) Patterns of contamination by organochlorines and mercury in the eggs of two river passerines in Britain and Ireland with reference to individual PCB congeners. *Environmental Pollution*, **76**, 233-243.

Ormerod, S.J., Tyler, S.J., & Jüttner, I. (2000) Effects of point-source PCB contamination on breeding performance and post-fledging survival in the dipper *Cinclus cinclus*. *Environmental Pollution*, **109**, 505-513.

Peakall, D.B. & Gilman, A.P. (1979) Limitations of expressing organochlorine levels in eggs on a lipid-weight basis. *Bulletin of Environmental Contamination and Toxicology*, **23**, 287-290.

Perrins, C.M. (1996) Eggs, egg formation and the timing of breeding. *Ibis*, **138**, 2-15.

Peterson, B.J. & Fry, B. (1987) Stable isotopes in ecosystem studies. *Annual Reviews Ecological Systems*, **18**, 293-320.

Phillips, D.L. & Gregg, J.W. (2001) Uncertainty in source partitioning using stable isotopes. *Oecologia*, **127**, 171-179.

Price, F.E. & Bock, C.E. (1983) Population ecology of the Dipper (*Cinclus mexicanus*) in the Front Range of Colorado. *Studies in Avian Biology No. 7*. Cooper Ornithological Society.

Rosing, M.N., Ben-David, M. & Barry, R.P. (1998) Analysis of stable isotope data: a K nearest-neighbors randomization test. *Journal of Wildlife Management*, **62**, 380-388.

Rounick, J.S. & Hicks, B.J. (1985) The stable carbon isotope ratios of fish and their invertebrate prey in four New Zealand rivers. *Freshwater Biology*, **15**, 207-214.

Strom, S.M., Ramsdell, H.S. & Archuleta, A.S. (2002) Aminolevulinic acid dehydratase activity in American dippers (*Cinclus mexicanus*) from a metal-impacted stream. *Environmental Toxicology and Chemistry*, **21**, 115-120.

Suedel, B.C., Boraczek, J.A., Peddicord, R.K., Clifford, P.A. & Dillon, T.M. (1994) Trophic transfer and biomagnification potential of contaminants in aquatic ecosystems. *Reviews of Environmental Contamination and Toxicology*, **136**, 21-89.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R. & Cushing, C.E. (1980) The river continuum concept. *Canadian Journal of Fisheries and Aquatic Science*, **37**, 130-137.

Won, H.T., Mulvihill, M.J. & Wakeford, B.J. (2000) Multi residue methods for the determination of chlorinated pesticides and polychlorinated biphenyls (PCBs) in wildlife tissues by gas chromatography/mass spectrometry. *Technical Report Series No. 335.* Canadian Wildlife Service, Environment Canada, Hull, Quebec.

Figure Legends

Figure 5.1. Arithmetic mean concentrations (± SE) of total organochlorines (OCs) and total polychlorinated biphenyls (PCBs) in μg/g wet weight detected in American dipper eggs from river and creek nests within the Chilliwack River watershed, British Columbia. Statistical tests were performed on log-transformed values (geometric means).

Figure 5.2. Stable isotope signatures ($\delta^{15}N$ and $\delta^{13}C$) in blood samples of American dippers and two reference species: mergansers and swallows with 95 % CI on both axes. Samples collected from the Chilliwack River watershed, British Columbia.

Figure 5.3. Stable isotope signatures ($\delta^{15}N$ and $\delta^{13}C$) of river resident and creek migrant American dippers in the Chilliwack River watershed relative to the mean isotopic signature (± SE) of benthic invertebrate and salmon fry diet sources.

Figure 5.4. Relative contributions of salmon fry to the diets of river resident and creek migrant dippers as determined by a two source linear mixing model (Phillips and Gregg 2001) using the ^{15}N isotope.

Figure 5.5. Stable isotope signatures ($\delta^{15}N$ and $\delta^{13}C$) in breast feather samples of resident and migrant American dippers in the Chilliwack River watershed, British Columbia.

Figure 5.6a and 5.6b. Correlation between $\delta^{15}N$ values in American dipper blood and contaminant concentrations (log $\sum$ DDT) and (log $\sum$ PCB) in egg samples. Blood and eggs are from individuals from the same territory in the same year (2001).

Figure 5.1

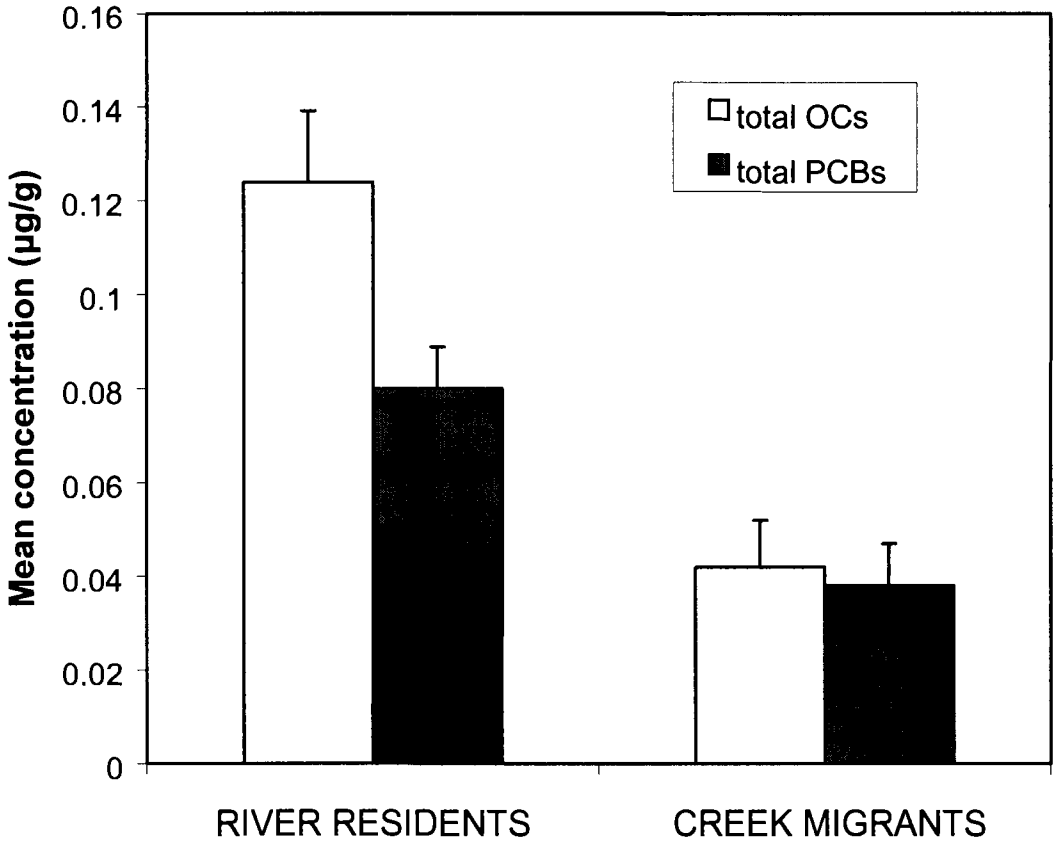
0.16
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
Mean concentration (µg/g)
total OCs
total PCBs
RIVER RESIDENTS
CREEK MIGRANTS

Figure 5.2

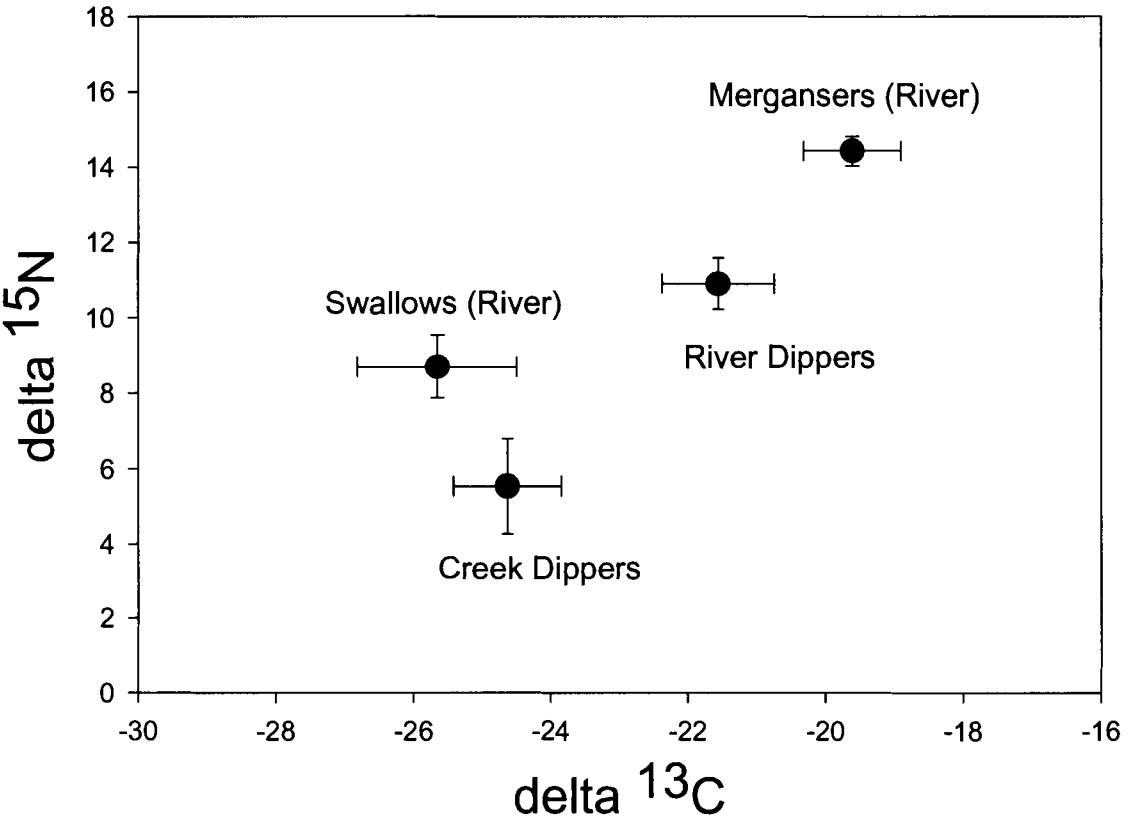
Mergansers (River)
Swallows (River)
River Dippers
Creek Dippers
delta 15N
delta 13C
18
16
14
12
10
8
6
4
2
0
-30
-28
-26
-24
-22
-20
-18
-16

Figure 5.3

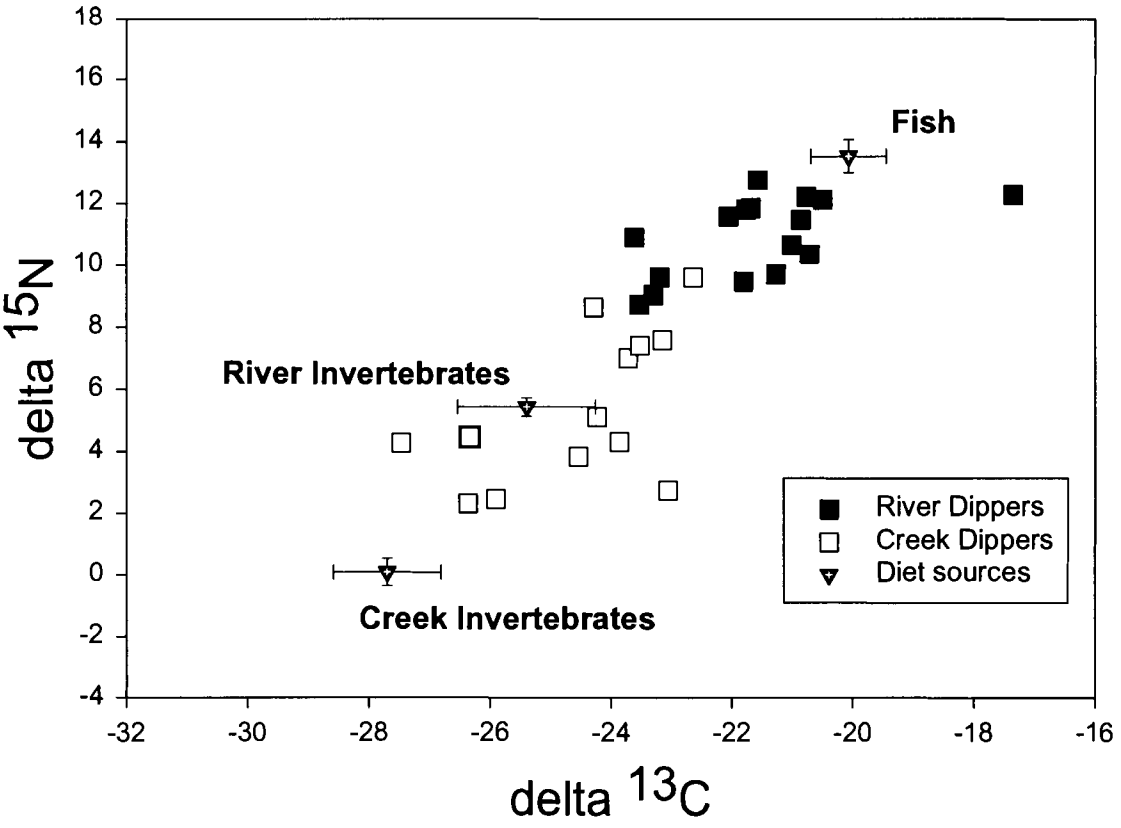
Fish
River Invertebrates
Creek Invertebrates
River Dippers
Creek Dippers
Diet sources
delta 15N
delta 13C
18
16
14
12
10
8
6
4
2
0
-2
-4
-32
-30
-28
-26
-24
-22
-20
-18
-16

Figure 5.4

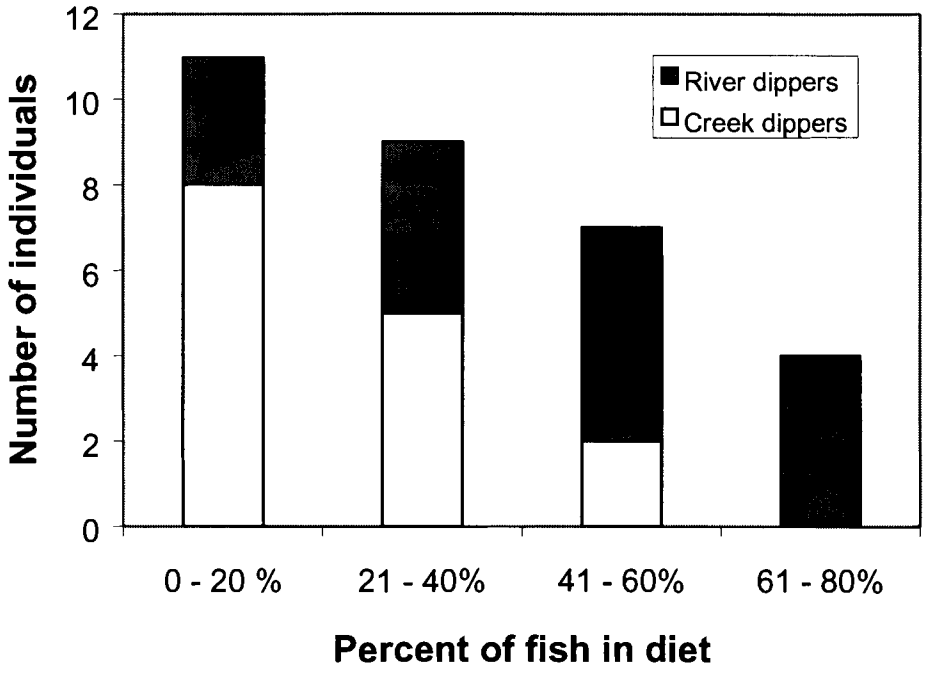
12
10
8
6
4
2
0
Number of individuals
River dippers
Creek dippers
0 - 20 %
21 - 40%
41 - 60%
61 - 80%
Percent of fish in diet

Figure 5.5

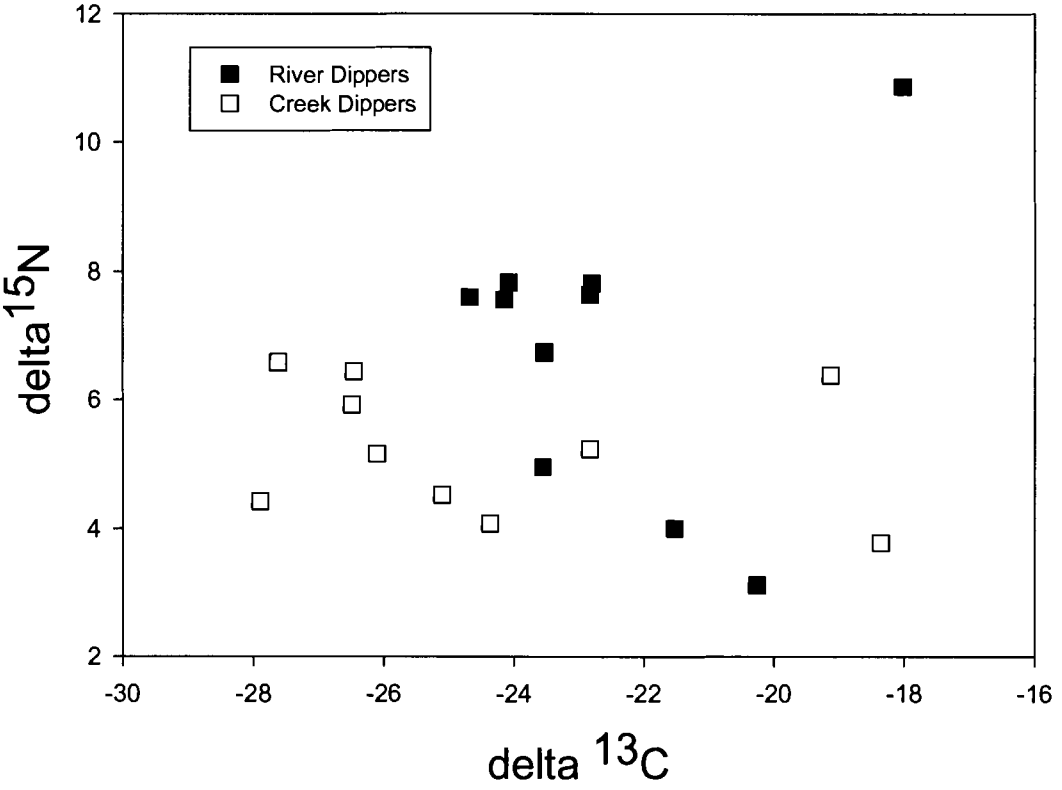
River Dippers
Creek Dippers
12
10
8
6
4
2
-30
-28
-26
-24
-22
-20
-18
-16
delta 15N
delta 13C

Figure 5.6a

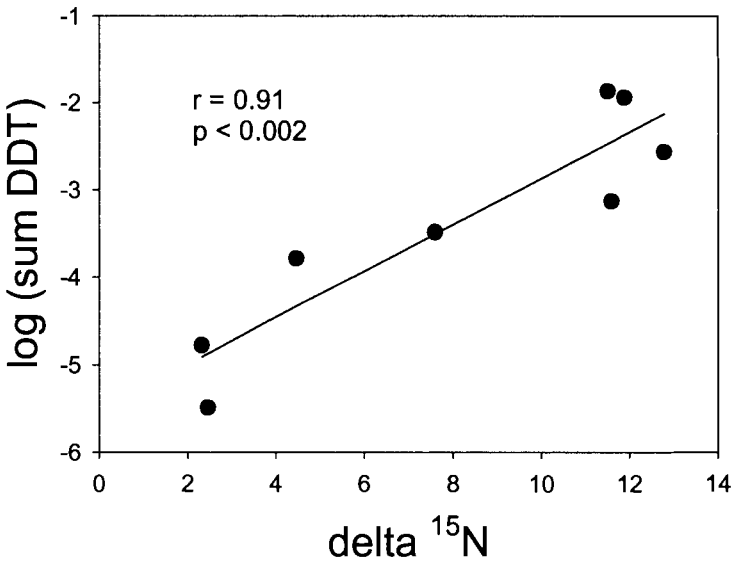

Figure 5.6b

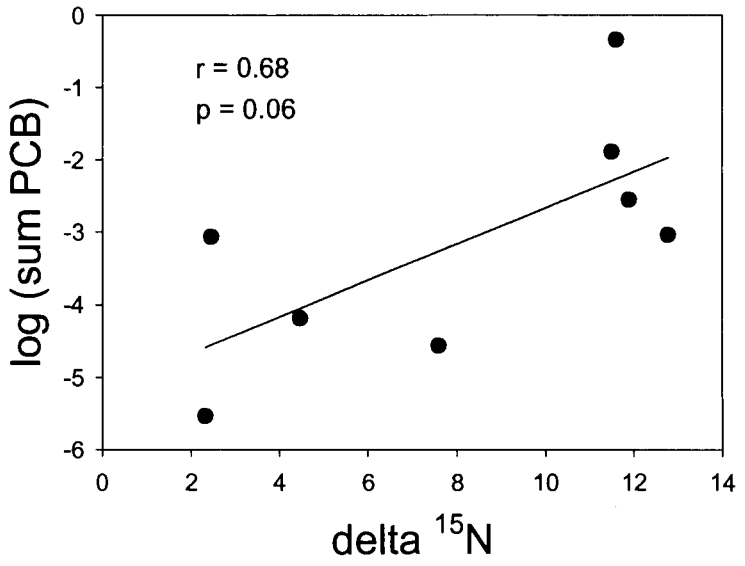

TABLE 5.1. Mean concentrations ± SE (geometric mean) of organochlorine residues detected in American dipper eggs collected from residents and altitudinal migrants in the Chilliwack River watershed.

Group	% Lipid	Concentration of Organochlorines (ng/g wet weight)[a]					
		∑ DDT	∑ ClBz	∑ CHLOR	∑ HCH	∑ Mirex	Dieldrin
River Resident	6.6 ± 0.4	118.0 ± 15.2 (102.2)	4.3 ± 0.5 (4.1)	2.4 ± 0.6 (1.5)	4.7 ± 1.9 (0.83)	ND[b]	ND
Creek Migrant	6.3 ± 0.3	42.6 ± 8.7 (29.9)	2.7 ± 0.5 (2.5)	0.7 ± 0.1 (0.42)	1.1 ± 0.6 (0.21)	ND	ND
Significance	NS	0.0002	0.006	0.009	0.06	-	-

[a]∑ DDT = DDE, DDT, DDD; ∑ ClBz = chlorobenzenes (tetrachlorobenzene, pentachlorobenzene, hexachlorobenzene); ∑ CHLOR = chlordane compounds (oxychlordane, *trans*-chlordane, *cis*-chlordane, *trans*-nonachlor, *cis*-nonachlor and heptachlor epoxide); ∑ HCH = hexachlorocyclohexanes (α- β- γ- hexachlorocyclohexanes); ∑ Mirex = mirex and photomirex

[b]ND = all samples non-detectable, detection limit for all compounds = 0.1 ng/g

TABLE 5.2. Mean concentrations (± SE) of mercury and selenium residues in resident and migrant American dipper eggs collected in the Chilliwack River watershed.

Group	% Moisture	Total Mercury (µg/g dry weight)[a]	Selenium (µg/g dry weight)
River Resident	75.7 ± 1.9	0.19 ± 0.01	2.96 ± 0.16
Creek Migrant	76.8 ± 1.6	0.09 ± 0.02	2.67 ± 0.19
Significance	NS	<0.0001	NS

[a] For samples below detection, a value of one-half the detection limit was used.
Hg detection limit = 0.18 µg/g

TABLE 5.3. Summary of mean isotopic values (± SE) for prey samples collected at eight sites on the main stem of the Chilliwack River and seven different tributaries.

Prey Type	n	$\delta^{15}N$ (‰)	$\delta^{13}C$ (‰)
Salmon fry	8	13.6 ± 0.5	-20.1 ± 0.6
River Invertebrates	8	5.4 ± 0.3	-25.4 ± 1.1
Creek Invertebrates	7	0.1 ± 0.4	-27.7 ± 0.9

TABLE 5.4. Summary of mean blood isotopic values (± SE) and relative diet composition (% fish obtained from linear mixing model for $\delta^{15}N$ isotope) of male and female American dippers in the Chilliwack watershed, British Columbia.

	Males (n = 20)	**Females (n = 11)**	**Significance (p)**
$\delta^{15}N$ (‰)	8.23 ± 0.75	8.48 ± 1.01	0.8 (NS)
$\delta^{13}C$ (‰)	-23.35 ± 0.47	-22.48 ± 0.64	0.3 (NS)
% Fish in diet	30.0 ± 4.9 %	37.4 ± 6.5 %	0.4 (NS)

CHAPTER 6:

ASSESSING EXPOSURE TO TRACE METALS IN MOUNTAIN STREAMS USING THE DIET, FEATHERS AND FECES OF THE AMERICAN DIPPER

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ABSTRACT

Trace metal deposition and mobilization from natural and anthropogenic sources threatens mountain streams across western North America by negatively affecting water quality. In the interest of developing a suitable biomonitor of metal pollution in watersheds, we examined trends in exposure to nine trace elements in the diet, feathers and feces of a partial migrant passerine, the American dipper (*Cinclus mexicanus*). We hypothesized that key differences may exist in exposure to metals for resident dippers on a river and altitudinal migrants on tributaries based on differences in prey metal levels between locations or possible differences in diet. Feathers of resident dippers on the main river had significantly higher concentrations of mercury, cadmium and copper relative to migrants on the tributaries. We used a mass balance approach to model metal contributions from different diets and breeding locations within the watershed on daily metal exposure to dippers while correcting for body mass. Considerable variability existed in the daily exposure to residents and migrants primarily due to differences in proportions of fish and invertebrates in the diet. In comparing actual metal exposure values to tolerable daily intake values for protection of wildlife consuming aquatic biota, we found most metals were below or within the range of tolerable daily intakes, except selenium, aluminum and zinc. Other metals such as cadmium, copper, and arsenic were only of concern for dippers feeding on primarily insect diets while mercury was of concern for dippers consuming mainly fish diets. These models were useful tools to demonstrate how shifts in diet and breeding location within a single watershed can result in changes in exposure that may be of toxicological significance.

Key words: *American dipper, metals, rivers, exposure models, ecological risk assessment, tolerable daily intake*

INTRODUCTION

Trace metals are present in aquatic systems world-wide largely from underlying substrates, natural erosion, volcanism and hydrological cycles. However, mining processes, urban and agricultural runoff, industrial emissions, and deforestation can also cause increased metal loads to watersheds (Nimmo et al. 1998, Scott et al. 2001, Arnason and Fletcher 2003, Miller et al. 2003). While mountain streams appear remote from industrialization and urbanization, many still contain significant concentrations of heavy metals from natural and anthropogenic sources (Gélinas and Schmit 1998, Lawson and Mason 2001). With concerns over environmental impacts of metals to freshwater ecosystems, it is important to be able to monitor the degree of metal exposure to organisms occupying mountain streams.

The American dipper (*Cinclus mexicanus*) is a potentially useful biomonitor of stream pollution because it is a year round resident of freshwater streams and has an exclusively aquatic diet comprised of benthic macroinvertebrates, small fish (salmon fry) and fish eggs. Many invertebrate taxa have the ability to bioaccumulate metals to high concentrations without inherent toxicity to the host species (Nehring 1976, Burrows and Whitton 1983). Freshwater fish are also known to bioaccumulate organometallic compounds particularly methylmercury (MeHg) due to the high assimilation efficiency and the slow elimination rates of this compound (Wiener and Spry 1996). Therefore, predators feeding on metal contaminated biota including the dipper are at risk for elevated exposure from its diet. Strom et al. (2002) determined adult and nestling American dippers were significantly exposed to lead through their invertebrate diet in a mine-impacted river system. Therefore, dippers may be an effective model for

monitoring metal pollution in mountain streams because they integrate contaminant sources from their aquatic diet over time and space.

Previous studies have indicated that the dipper has distinct altitudinal patterns of migration, which includes seasonal movement upstream and downstream within a watershed (Morrissey et al. submitted-a). Resident and altitudinal migrant dippers shared common wintering grounds on the river, but most migrants moved upstream onto creeks in the spring while residents remained on the river to breed. Previous work in a coastal watershed of British Columbia, Canada, indicated that river residents had elevated levels of mercury and chlorinated hydrocarbons in eggs relative to creek migrants as a result of atmospheric deposition and differences in the proportion of fish and invertebrates in the diet. (Morrissey et al. submitted-b). Proportions of fish in the diet varied from 0 % to 71 % with river residents consuming significantly more fish (42 %) compared to creek migrants (22 %). Therefore, we hypothesized that trace metal concentrations in the feathers and feces of resident and migrant dippers may also reflect the bird's migratory status or its specific diet.

Measuring metal levels in excretory tissues such as feathers and feces is now a common tool to examine environmental pollution that is non-injurious and non-invasive to birds. Metal excretion is generally through feces or by deposition in the uropygial gland, salt gland, and into feathers during molt (Goede and de Bruin 1984, Burger 1993, Burger and Gochfeld 1985, Dauwe et al. 2000). For females, egg laying can also be a route of excretion for several metals (Burger and Gochfeld 1993, Burger 1994, Dauwe et al. 1999). While many studies are able to determine food chain contamination by

sampling excretory tissues such as feathers, feces and eggs; few studies verify how levels in these excretory tissues relates to exposure from the diet.

Our main objective in this study was to determine if any differences exist in metal exposure for resident dippers occupying the main river and migrant dippers breeding on watershed tributaries using feathers and feces as bioindicators. Furthermore, we aimed to determine the major sources of metal contamination to resident and migrant dippers via the fish and invertebrate diet in order to quantify the degree of exposure and relate this to levels observed in feathers. Although some elements are biologically essential, all are toxic at high enough concentrations; with some having a very narrow window of essentiality and toxicity (Ash and Stone 2003). Therefore, in the interest of using the dipper as a biomonitor, we modeled the potential toxicological risks of metal exposure to dipper populations with different migratory strategies and with different diets.

METHODS

Collection of samples

Aquatic larval invertebrates (~ 1 g dry weight) were collected either by kick sampling in the stream (disturbing the rocks directly upstream of a Surber sampler) or by turning over rocks by hand. The sample represented a mixture of insect taxa that dippers would naturally prey upon including approximately equal proportions of Ephemeropteran, Plecopteran and Tricopteran larvae in addition to a much smaller fraction by mass of Coleopteran and Dipteran larvae. Up to ten individual salmon fry (*Oncorhynchus* spp.) (age-0) that each weighed 100-200 mg fresh weight, were captured live from the river using a dip net and represented a composite sample of predominantly coho and chum salmon fry (~80%) but pink and chinook salmon (~20%) were also

included. Composite samples of benthic invertebrates and salmon fry were collected at eight different sites along the main stem of the river. An additional seven composite samples of invertebrates were also collected from seven different tributaries in the watershed. All samples were collected during a one-week period in late April 2000 and 2001 prior to the spring freshet. Samples were subsequently washed three times with distilled deionized water to remove any surface contamination or stream water and stored frozen in acid rinsed glass vials until preparation for trace metal analysis.

Several breast feathers were plucked from individual adult birds at the time of capture and banding for metal analysis. The sex of each bird was obtained at the time of capture using wing chord measurements (Kingery 1996). Through the use of color banding, only birds of known migratory status (river resident or creek migrant) were used for the metal analysis. Each individual sample of 7-10 feathers was stored in polyethylene bags and refrigerated until analyses. We used 105 feather samples for Hg analysis and 80 of these were also analyzed for additional multiple elements. Fecal samples (n =14) were collected from nestlings (n = 5) opportunistically during banding of chicks at 12-14 days of age or by following adults (n =9) and collecting fresh feces off of rocks, with care taken to avoid contamination from the substrate. Fecal samples were stored frozen in acid washed plastic containers until analysis.

Sample preparation and metal analysis

Methods for sample preparation and digestion were adapted and modified from Canadian Wildlife Service method MET-CHEM-AA-02 (Neugebauer 2000) and U.S. Environmental Protection Agency method 200.3 (McDaniel 1991). Feathers were washed with pure acetone, 1% Triton-X solution alternated with several rinses of distilled

deionized water to remove any external surface contamination. Samples were then air dried for 48 hours and finally oven dried for 12 hours. Invertebrate, fish and fecal samples were freeze-dried for 24- 48 hours until constant weight was achieved. Samples were then accurately weighed into acid washed glass flasks to the nearest 1 mg. The digestion procedure involved adding 5 mL of 70% ultrapure nitric acid (HNO_3), slow heating to reduce volume, adding an additional 2 mL HNO_3 while heating and finally 1 mL of 30 % ultrapure hydrogen peroxide (H_2O_2). All samples were reduced by heat to < 1 mL and diluted to 10 mL with distilled deionized water, then stored in 15 mL polypropylene vials refrigerated until metal analysis. A minimum of two certified reference materials (Dolt-2 and Tort-2) (National Research Council Canada) and two procedural blanks were digested simultaneously with every batch of samples and analyzed for quality assurance. In addition, a standard calibration curve, analytical blanks and spiked samples were run with each analysis.

Metal analysis was performed using an inductively coupled plasma mass spectophotometer (ICP-MS) (Levelton Engineering, Richmond, B.C.) for feathers and fecal samples or an inductively coupled plasma atomic emission spectrophotometer (ICP-AES) (Cavendish Analytical Laboratories, Vancouver, B.C.) for invertebrates and fish. Over 25 different elements were obtained from these analyses but we report only the data for mercury (Hg), cadmium (Cd), lead (Pb), selenium (Se), manganese (Mn), copper (Cu), zinc (Zn), aluminum (Al), and arsenic (As) herein after referred to as metals. All metal concentrations are expressed in μg/g dry weight (ppm). Recoveries of reference materials were within 10 % of the certified values or were recovery corrected if outside this range (invertebrates and fish from 2000 only).

Data analysis

Both arithmetic and geometric mean concentrations of metals in the diet, feathers and feces were calculated and reported ± SE to facilitate comparison with other studies. In addition, we report the proportion of samples detected for each metal as a measure of prevalence. Metal concentrations generally exhibited a non-normal distribution (Shapiro-Wilk W test) and were therefore log transformed to improve normality prior to performing statistical comparisons. We used a two-way ANOVA followed by a Tukey multiple comparison procedure to compare the metal concentrations among river invertebrates, creek invertebrates and fish by year. A three-way ANOVA (generalized linear model) was used to analyze feathers for effects of migratory group: resident (n = 40) and migrant (n = 40), collection year (1999, 2000, 2001) and sex (male or female) as well as interaction terms. Given the limited number of fecal samples (n =14), the power for statistical comparisons was weak and is therefore only reported as means of all samples. Pearson product moment correlation coefficients were used to test for correlations among metal concentrations in both feathers and feces. Statistical tests were performed using JMP IN v. 4.0 (SAS Institute) and the significance level was set at α = 0.05.

Exposure models

A mass balance approach was used to calculate daily metal exposure to American dippers depending on the breeding location within the watershed (river or creek) and the relative contributions of fish and invertebrates to the diet. Models incorporated geometric mean metal concentrations detected in invertebrates and fish collected from the main river and tributaries of the Chilliwack watershed in addition to estimated daily

intake of each prey item using published dipper energy requirements. Given the importance of body mass in comparing daily exposure among species (Sample et al. 1996), we further corrected the daily ingestion value for average dipper body mass. We assumed that the primary route of exposure would be through oral ingestion of the diet items. However, some metals may be taken up through the water directly by drinking but this was not accounted for. Therefore, our model is considered a conservative estimate of exposure to metals. The exposure model for American dippers in the Chilliwack River watershed was therefore calculated as follows:

$$E_{metal} = \frac{(W_f \times C_f) + (W_i \times C_i)}{BW}$$

where E_{metal} = exposure to metal x (µg/g body weight per day), W_f = weight of fish eaten per day (g/day), C_f = geometric mean concentration of metal in fish (µg/g), W_i = weight of invertebrates eaten per day (g/day), C_i = geometric mean concentration of metal in invertebrates (µg/g), BW = body weight of dipper (mean = 55 g). Weight of fish (W_f) and invertebrates (W_i) eaten on a daily basis were calculated using the following equations:

$$W_f = ((P_f \times DEE) \times AE) \times ED_f$$

$$W_i = ((P_i \times DEE) \times AE) \times ED_i$$

where P = proportion of fish or invertebrates in the diet, DEE = average daily energy required by dippers (estimated ~ 48.04 kcal/day) (Bryant and Tatner 1988), AE = assimilation efficiency correction factor for fish diet (85% or 1.15) or invertebrate diet (70% or 1.3), ED = energy density of juvenile salmon (5.7 kcal/g dry weight) (Higgs et al. 1995) or aquatic invertebrates (4.8 kcal/g dry weight) (Cummins and Wuychuk 1971). These estimates for amount of food eaten per day closely matched the allometric equation of daily food ingestion rate for passerines FI = 0.398 ($BW^{0.85}$) given by Nagy (1987).

The allometric equation estimates daily food ingestion of a 55 g dipper to be 12.0 g/day dry weight while our estimates using daily energy requirements averaged 11.8 g/day dry weight.

Each metal exposure model was compared to a tolerable daily intake (TDI) calculated using the Canadian Tissue Residue Guidelines for the Protection of Wildlife Consumers of Aquatic Biota protocol (CCME 1998). The TDI is calculated from the results of avian chronic toxicity tests in which the substance was administered orally and sensitive endpoints were measured (Appendix 1). TDI is calculated using the geometric mean of the NOAEL and the LOAEL and dividing by an uncertainty factor to account for differences between species (usually 10):

$$TDI = (LOAEL \times NOAEL)^{0.5} \div UF$$

where:

TDI = tolerable daily intake

LOAEL = lowest-observed-adverse-effect-level

NOAEL = no-observed-adverse-effect-level

UF = uncertainty factor

The NOAEL and LOAEL for suitable avian toxicity tests were taken from the literature and summarized by Sample et al. (1996). The TDI value is in units of μg/g body weight/day for direct comparison with the values in the exposure model for American dippers.

RESULTS

Metals in diet items: invertebrates and fish

Invertebrate samples from the river and the tributaries generally did not differ significantly in metal concentrations (Table 1). Copper was the only metal found to be significantly higher in the river invertebrates relative to the creeks (t_{28}= -2.45, p = 0.02), although Cd, Pb, Mn and Zn also showed similar patterns to Cu. Significant differences existed in metal concentrations between invertebrate and fish samples. In all cases except for Hg and Pb, fish had lower concentrations of metals than both the river and creek invertebrate samples ($p < 0.0001$) (Table 1). For Hg, fish concentrations were almost 4 times higher and were detected more frequently (47 %) than invertebrates (20 %). For Pb, there was no difference in residue levels between fish and invertebrate samples but lead was detected at a higher frequency (100%) in fish relative to invertebrates (70%). There was no effect of collection year for the majority of metals in invertebrates and fish. Only Se, Zn and As were significantly lower ($p < 0.0001$) for invertebrates collected in 2000 relative to 2001.

Metals in feathers and feces

Metals most commonly detected in dipper feather samples in decreasing order were Zn> Cu> Hg> Se> Pb> Mn> Cd> Al> As. Migratory status (river resident or creek migrant) was significant in predicting Hg, Cd and Cu feather concentrations with river residents having higher levels of these metals than the creek migrants (Table 2). Aluminum (n = 13) and As (n = 1) were not frequently detected in dipper feathers. Feathers collected from adult dippers were further analyzed using a 3-way ANOVA with interaction terms to determine the effects of migratory group, sex and year on metal

concentrations. No interaction terms were significant and were therefore removed from the analysis. There was an effect of sex on Mn ($p = 0.002$) and Zn ($p = 0.03$) concentrations with Cu being marginally insignificant ($p = 0.06$). In all cases, females tended to have higher feather metal concentrations relative to males. The year of collection was important for predicting Hg levels ($p<0.0001$) and Mn levels ($p = 0.03$). In general, levels were higher in 1999 for Hg and higher in 2001 for Mn. Migratory status was only an important effect in predicting higher Hg and Cd feather concentrations in resident feathers when corrected for the other variables.

Mean metal concentrations for adult and nestling fecal samples were not significantly different and were therefore pooled and reported as a single value (Table 3). There was no difference in metal concentrations by sex or migratory status; however, due to small sample sizes, statistical power was extremely limited. Metals were detected in 100% of the fecal samples analyzed, with the exception of one low weight sample in which mercury was not detected. For all metals except for Se, the fecal concentrations (geometric means) exceeded those in the invertebrate and fish prey items.

Few metals ($\log_{10}$) in dipper feathers showed significant positive correlations; however, all were weak ($r \leq 0.35$) and bordered significance. Stronger correlations were found between several metals in the feces, including Mn with Al ($r = 0.80$, $p = 0.0007$) as well as for Hg and Se ($r = 0.73$. $p = 0.005$). Other correlations in fecal samples included As and Al ($r = 0.70$, $p = 0.006$), Hg and Zn ($r = 0.66$, $p = 0.01$), Mn and As ($r = 0.66$, $p = 0.01$), and Se with Cd ($r = 0.56$, $p = 0.04$). Only Hg and Zn were significantly correlated in both the feathers and feces.

Exposure assessment

In modeling the degree of daily metal exposure to dippers in the Chilliwack watershed, two general trends emerged. First, for birds breeding on the river (residents), the predicted exposure for Cd, Cu, Pb and Zn was generally higher than for those birds breeding on tributaries (migrants) (Fig.1 a-d). For Se, Al, As, Mn and Hg, there was very little or no differences between river and creek locations, or creek values were slightly higher (Fig. 2 a-e). The second major trend in the exposure models was the effect of diet on metal concentrations, which generally exceeded the effect of watershed breeding location (migratory status). Specifically, increasing proportions of fish relative to invertebrates in the diet resulted in a decrease in metal concentrations. Therefore, for all metals except Hg, there was a negative relationship between proportion of fish in the diet and calculated metal exposure (Figs. 1 and 2). Therefore, peak daily exposure of most metals occurred for restricted invertebrate feeders. In the case of Hg, the relationship was reversed; demonstrating an increase of fish in the dipper's diet produced an increase in Hg exposure (Fig. 2e).

Each exposure model for the metals of interest was compared to a tolerable daily intake (TDI) level estimated to protect wildlife consumers of aquatic biota (CCME 1998) (Appendix 1). Calculated daily metal exposures exceeded TDIs for selenium, Al and Zn. Both Pb and Mn were well below TDI levels. Of further interest were those metals that partly exceeded the TDI depending on the individual's diet or location or both. These metals included Cd, Cu, As and Hg. Cadmium, Cu and As exposure only exceeded the TDIs for dippers feeding mainly on invertebrates. Mercury exposure was generally greater for birds feeding on fish assuming Hg in the diet was primarily MeHg.

DISCUSSION

Significance of feather and fecal metal levels with respect to diet

Through analysis of prey items, feathers and feces, we found American dippers in the Chilliwack River watershed were exposed to a suite of trace metals including selenium, copper, zinc, aluminum, mercury and cadmium. Although this watershed is not impacted by mining or other discharge point sources, several elements may be mobilized as a result of natural hydrological processes, soil erosion from deforestation or long-range transport and atmospheric deposition. Therefore, the American dipper appears to be an excellent model for monitoring metal pollution in mountain streams.

Several studies have determined that Hg, Pb and Cd among other metals are significantly deposited in the feather during the period of feather growth, producing metal profiles that remain inert and stable (Myack et al. 1981, Applequist 1984, Burger 1993, Dauwe 2002). Concentrations of metals in the feathers reflect levels in blood at the time of feather growth, either from current dietary sources or from mobilization of metals in internal organs (Burger 1993). Feathers are therefore a useful means of non-lethal sampling to evaluate metal burdens in aquatic and terrestrial birds. We anticipated that breeding location as a result of migratory status would influence metal levels in feathers. For Hg, Cd and Cu, migratory status was significant in predicting feather profiles with river residents having higher concentrations than creek migrants. This could be attributed to either differences in metal concentrations between the river and creek breeding locations or differences in the diet between residents and migrants (proportion of fish and invertebrates).

From the exposure models, we anticipated that migrants would have higher exposure to most metals (except Hg) primarily because of the larger proportion of invertebrates in their diet. However, residents were found to have significantly higher concentrations of Cu and Cd in their feathers. Copper was found to be significantly higher in river invertebrates compared to creek samples, but Cd also followed similar trends. Therefore, resident birds occupying the main river year round may be exposed to higher concentrations of Cu and Cd from their invertebrate prey, thus influencing metal profiles in feathers. For Hg, no differences were detected in invertebrate samples between locations, implying it may be caused by diet differences. In contrast to other metals, Hg is of greater importance to aquatic birds on primarily fish diets due to the prevalence of the more toxic MeHg in fish tissue (Wiener and Spry 1996). Fish were found to have almost 4x higher Hg concentrations and a higher frequency of detection than invertebrates. In our previous work, Hg levels in the eggs of river residents were also found to be higher than migrants (Morrissey et al. submitted-b). Furthermore, the highest Hg levels detected in dipper feathers (2.74 μg/g and 2.09 μg/g) were from a resident pair at the fish hatchery. Since resident dippers are known to consume greater proportions of fish than migrants (Morrissey et al. submitted-b), diet was likely more important in predicting higher Hg exposure and consequently higher feather Hg levels.

In modeling the effects of food chain differences on daily metal exposure, we found that the influence of diet generally exceeded that of breeding location. Species that eat prey from different levels in the food chain have contaminant levels that are strongly influenced by diet (Hebert et al. 1997, Bustnes et al. 2000, Bearhop et al. 2000). Food chain differences among marine birds were important in explaining variation in metal

concentrations in eggs (Burger 2002) and tissues (Elliott et al. 1992). With the exception of Hg, invertebrates from the Chilliwack watershed had higher concentrations of all metals, placing insectivorous wildlife at greater risk to increased metal uptake. Mercury exposure, in contrast, is greater for piscivorous wildlife. Therefore, resident American dippers consuming more fish are generally at greater risk for exposure to Hg, while migrants feeding on mainly insects were predicted to have greater exposure to other metals.

Sex had little effect on feather levels although females were found to excrete greater amounts of Mn and Zn in the feathers. Few studies have reported gender differences in feather profiles; however, Burger and Gochfeld (1992) found Pb and Cd levels were also higher in female Black skimmer (*Rynchops niger*) feathers despite the fact that males are larger and may eat larger and more contaminated prey. We have no evidence to suggest why female dippers had higher feather Mn and Zn since males and females do not have different diets (Morrissey et al. submitted-b).

Feathers have been used since the 1960s for indicating metal exposure in birds; however, recent studies show excretement can also be a sensitive indicator of metal contamination in birds (Fitzner et al.1995, Spahn and Sherry 1999, Dauwe et al 2000). Metals were detectable in almost all fecal samples collected at concentrations exceeding those of the prey items. Spahn and Sherry (1999) also found that Little Blue heron (*Egretta caerulea*) fecal samples contained higher concentrations of metals than their prey and suggested this was because feces largely represent the unabsorbed remnants of multiple food items. In this study, sample size limitations restricted inferences from the data, but nevertheless they provide a baseline for future studies on this species and other

river birds. Since metals found in feces are readily detected, often at higher concentrations than the diet items, they can provide a non-destructive and quantifiable means of monitoring food chain contamination from trace metals.

Correlations among metal levels for the feathers and feces showed only a small number of significant relationships and only Hg was significantly correlated with Zn in both feathers and feces, indicating no clear patterns with respect to metal excretion mechanisms. However, key correlations among metals in fecal samples may be important for understanding the kinetics and toxicity of metals in dippers. Aluminum and Mn were positively correlated suggesting similar metal availability or metabolism. Both elements are earth metals that are typically derived from natural mineral deposits. They were excreted in high concentrations in feces, indicating they are either abundant in this system or not readily bioavailable to the birds. Mercury was also significantly correlated with Se in fecal samples. It is known that inorganic Hg is often bound to Se in liver and other tissues, and that Hg and Se interact to counter the toxicity of each other (Cuvin-Aralar and Furness 1991). Given the high levels of Se detected in invertebrate and dipper samples and the correlation in feces, Se and Hg may be interacting to produce ameliorative effects to dippers.

Tolerable daily intakes and toxicity concerns

Many bird populations may be subject to the effects of chronic exposure to low-level toxicants resulting in reproductive dysfunction, increased susceptibility to disease or other stresses, and changes in normal behavior (Scheuhammer 1987). Although many bird populations are chronically exposed to low levels of metals through the diet, it is difficult to determine critical threshold levels relevant to all species. In particular, there

are relatively few controlled laboratory studies examining the effects of toxic metals on passerines. In addition, since dippers belong to the unique family *Cinclidae*, the world's only truly aquatic passerines, direct comparisons of toxicity tests from other species may be inappropriate. For this reason, we selected the approach of determining a tolerable daily intake value, which included a marginal uncertainty factor. The TDI represents a recommended safe daily intake level for this species (based on body weight and daily dose) that should not cause any sublethal effects to populations. The only metals to which dippers on any diet clearly exceeded the TDI guidelines were Se, Al, and Zn. Those elements are either homeostatically controlled or are essential elements where the range of essentiality and toxicity is not well understood. However, mortality and reproductive effects from Se exposure, particularly to aquatic birds in areas receiving agricultural drainage, have been documented (Olendorf et al. 1988, Olendorf et al. 1989). Aluminum has also been reported to influence reproduction of insectivorous passerines breeding in acid sensitive environments (Nyholm 1981, Miles et al. 1993). For Zn, evidence of toxicity to wild birds is limited primarily because Zn is internally regulated even when birds are exposed to high levels of contamination (Burger 1993, Janssens et al. 2002). Both Pb and Mn were well below TDI values and were not elevated in feather samples. Lead and Mn, which are found in environmental samples primarily from combustion of gasoline additives (Cooper 1984), were not significantly elevated in prey from this watershed and were not at levels known to cause toxicity in birds (Laskey and Edens 1984, Eisler 1988).

Excess Se exposure through the diet particularly in the form of selenomethionine is known to cause reproductive failure and mortality in birds (Heinz 1996). Food chain

organisms such as benthic invertebrates and fish can accumulate high concentrations of Se without toxicity to the host; however, a dietary toxicity threshold for fish and wildlife is recommended at 3 μg/g dry weight (Lemly 1996). Since all the invertebrate prey samples and many fish samples collected from the Chilliwack watershed exceeded this guideline, there may be cause for concern. Heinz et al. (1989) showed that mallard (*Anas platyrhynchos*) reproduction was impaired when dietary concentrations of selenomethionine were between 4 and 9 ppm. Although we do not have any information about the concentrations of the more toxic organic form of Se (selenomethionine) in dipper prey, daily Se exposure was 6x higher than the TDI levels for birds on exclusively invertebrate diets. Feather concentrations were not different among migratory groups, but Se levels were higher in invertebrate samples compared to fish indicating migrant dippers on mainly invertebrate diets may be at a higher risk to toxic effects from Se.

Several orders of aquatic invertebrates including chironomids, caddisflies, stoneflies and mayflies have been shown to exhibit high Al concentrations of 0.1- 0.3 % body weight (dry weight) (Sadler and Lynam 1985, Ormerod et al. 1988). Insectivorous birds feeding on contaminated invertebrates may be exposed to sufficiently high levels of Al to cause reproductive effects seen in other species, particularly if calcium (Ca) and phosphorus (P) are limiting (Nyholm and Myhrberg 1977, Nyholm 1981, Sparling 1990). Invertebrates sampled in 2001 from the Chilliwack watershed had elevated Al levels in the range of 0.05 – 0.43 % (mean = 0.12 % dry weight). Aluminum is of particular concern in acid sensitive regions, especially in ecosystems with exposed granite or other calcium poor substrates, which are most severely affected by acidification (Schindler et al. 1989). The Chilliwack watershed, in addition to many similar river basins in the

region, is largely composed of granitic bedrock, making this system vulnerable to solubilization of metals that are more readily bioavailable to aquatic biota. Swain (1987) listed the Chilliwack Lake, a source at the headwaters of the Chilliwack River, as one of 20 % of British Columbia lakes with high sensitivity to acid inputs using measures of pH, calcium and alkalinity. Acidified environments generally expose wildlife to increased dietary Al, Cd, Pb and Hg (Scheuhammer 1991). Although Al was not readily detected in all dipper feather samples and was excreted in high concentrations in the feces, future studies to determine whether acid deposition is contributing to observed Al levels are warranted.

Other metals including Cd, Cu, and Hg were found to exceed tolerable daily intakes depending on individual diet or location or both. Cadmium and Cu exposure in particular was greater for birds on the river eating mainly invertebrates. There is concern because those elements are known to accumulate in target organs (kidneys) in excess of the levels in the food supply (Hunter and Johnson 1982, Scheuhammer 1987, Larison et al. 2000). For example, with continued low-level dietary Cd exposure, there is a persistent increase in renal Cd with very little excretion. Long-term exposure can eventually result in high kidney Cd levels that could reach a critical concentration of 100-200 μg/g causing renal tubular necrosis (Scheuhammer 1987). Other sublethal effects on immature and adult birds may be apparent at lower concentrations. Therefore, monitoring exposure to bioaccumulative elements such as Cd in insectivorous birds remains important for many watersheds.

In general, Hg levels detected in this study were below reported toxic thresholds for birds. However, predicted daily exposure was considerably greater for birds

consuming high fish diets, thus exceeding the TDI. Consistent with the model, resident dippers had higher feather Hg concentrations, but within the range of 1-5 ppm, considered as background exposure (Scheuhammer 1987). It is unclear whether dippers are unique in their sensitivity to Hg. Heinz (1979) found that dietary levels of 0.5 μg/g dry weight MeHg were significant to cause female mallards to lay fewer eggs and produce fewer ducklings in addition to behavioral changes in ducklings. Barr (1986) found reductions in egg laying and nest site and territory fidelity in Common loons (*Gavia immer*) on diets containing 0.2 to 0.3 μg/g wet weight. Mercury concentrations detected in dipper prey were below those values, but dippers consuming primarily fish diets exceeded TDI levels, which account for interspecies differences in body mass and total daily exposure. Furthermore, TDI values used in this study were set to maintain protection to the most sensitive species consuming diets of MeHg. Given that nearly 95 to 99 % of mercury in fish is MeHg (Bloom 1992) and this is the most important source to dippers, we believe that TDI values for Hg used in this study should be considered more appropriate than direct comparison of diet concentrations.

Summary

Modeling the effects of diet and location to estimate daily metal exposure is an effective way to initially assess exposure risks to wildlife that have a varying diet or migratory strategy. Exposure models can then be compared to tolerable daily intakes, which is arguably a more useful method for determining exposure risks than simply comparing metal concentrations in the diet items to known toxic concentrations. Several studies have shown the effects of body size on physiological functions including metabolic rate and responses to toxic chemicals (Sample et al. 1996). Therefore, we

recommend using specific exposure models to directly assess which contaminants are of concern by location and species given they more accurately account for daily exposure from all diet items while controlling for the effects of body mass. Future work should involve additional modeling to attempt to predict tissue residues from dietary exposure and to estimate assimilation efficiency of trace metals from daily ingestion and fecal deposition.

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REFERENCES

Applequist, H., Asbirk, S., Drabaek, I. 1984. Mercury monitoring: Mercury stability in bird feathers. Mar Pollut Bull 15: 22-24.

Arnason, J.G., Fletcher, B.A. 2003. A 40+ year record of Cd, Hg, Pb, and U deposition in sediments of Patroon Reservoir, Albany County, NY, USA. Environ Pollut 123:383-391.

Ash, C., Stone, R. 2003. A question of dose. Science 300: 925.

Barr, J.F. 1986. Population dynamics of the Common loon (*Gavia immer*) associated with mercury-contaminated waters in northwestern Ontario. Occassional Paper No. 56. Canadian Wildlife Service. Ottawa, Ontario. 23 pp.

Bearhop, S., Waldron, S., Thompson, D., Furness, R. 2000. Bioamplification of mercury in the Great skua *Catharacta skua* chicks: the influence of trophic status as determined by stable isotope signatures of blood and feathers. Marine Pollut Bull 40: 181-185.

Bloom, N.S. 1992. On the chemical form of mercury in edible fish and marine invertebrate tissue. Can J Fish Aquat Sci 49: 1010-1017.

Bryant, D.M., Tatner, P. Energetics of the annual cycle of Dippers *Cinclus cinclus*. Ibis 130: 17-38.

Burger, J. 1993. Metals in avian feathers: Bioindicators of environmental pollution. Rev Environ Toxicol 5: 203-311.

Burger, J. 1994. Heavy metals in avian eggshells: another excretion method. J Toxicol Environ Health 41:207-220.

Burger, J. 2002. Food chain differences affect heavy metals in bird eggs in Barnegat Bay, New Jersey. Environ Research Section A 90: 33-39.

Burger, J., Gochfeld, M. 1985. Comparison of nine heavy metals in salt gland and liver of great scaup (*Aythya marila*), black duck (*Anas rubripes*) and mallard (*Anas platyrhynchos*). Comp Biochem Physiol 81: 287-292.

Burger, J., Gochfeld, M. 1993. Lead and cadmium accumulation in eggs and fledgling seabirds in the New York bight. Environ Toxicol Chem 12: 261-267.

Burrows, L.G., Whitton, B.A. 1983. Heavy metals in water, sediments and invertebrates from a metal-contaminated river free of organic pollution. Hydrobiologia 106: 263-273.

Canadian Council of Ministers of the Environment (CCME). 1998. Protocol for the derivation of Canadian tissue residue guidelines for the protection of wildlife that consume aquatic biota. Canadian Council of Ministers of the Environment, Winnipeg, Manitoba. [Reprinted in Canadian environmental quality guidelines, Chapter 8, CCME, 1999, Winnipeg.]

Cooper, W.C. 1984. The health implications of increased manganese in the environment resulting from the combustion of fuel additives: a review of the literature. J Toxicol Environ Health 14: 23-46.

Cummins, K.W., Wuycheck, J.C. 1971. Caloric equivalents for investigation in ecological energetics. Mitt Intern Ver Limnol 18: 1-158.

Cuvin-Aralar, M.L.A., Furness, R.W. 1991. Mercury and selenium interaction: a review. Ecotoxicol Environ Saf 21: 348-364.

Dauwe, T., Bervoets, L. Blust, R., Pinxten, R., Eens, M. 1999. Are eggshells and egg contents of great and blue tit suitable as indicators of heavy metal pollution? Belg J Zool 129: 439-447.

Dauwe, T., Bervoets, L. Blust, R., Pinxten, R., Eens, M. 2000. Can excretement and feathers of nestling songbirds be used as biomonitors for heavy metal pollution? Arch Environ Contam Toxicol 39: 541-546.

Dauwe, T., Bervoets, L. Blust, R., Eens, M. 2002. Tissue levels of lead in experimentally exposed zebra finches (Taeniopygia guttata) with particular attention on the use of feathers as biomonitors. Arch Environ Contam Toxicol 42: 88-92.

Elliott, J.E., Scheuhammer, A.M., Leighton, F.A., Pearce, P.A. 1992. Heavy metal and metallothionein concentrations in Atlantic Canadian seabirds. Arch Environ Contam Toxicol 22: 63-73.

Eisler, R. 1988. Lead hazards to fish, wildlife and invertebrates: a synoptic review. U.S. Fish and Wildlife Service. Biological Report No. 85 (1.1).

Fitzner, R.E., Gray, R.H., Hinds, W.T. 1995. Heavy metal concentrations in Great Blue heron castings in Washington State: a technique for monitoring regional and global trends in environmental contaminants. Bull Environ Contam Toxicol 55: 398-403.

Gélinas Y., Schmit J.P. 1998. Estimation of the bulk atmospheric deposition of major and trace elements to a rural watershed. Atmospheric Environment 32: 1473-1483.

Goede, A.A., de Bruin, M. 1984. The use of bird feather parts as a monitor for metal pollution. Environ Pollut Ser B 8: 281-298.

Hebert, C.E., Shutt, J.L., Norstrom, R.J. 1997. Dietary changes cause temporal fluctuations in polychlorinated biphenyl levels in Herring gull eggs from Lake Ontario. Environ Sci and Technol 31: 1012-1017.

Heinz G.H. 1979. Methyl mercury: Reproductive and behavioral effects on three generations of mallard ducks. J Wild Manage 43: 394-401.

Heinz, G.H., Hoffman, D.J., Gold, L.G. 1989. Impaired reproduction of mallards fed an organic form of selenium. Environ Toxicol Chem 6: 423-433.

Heinz, G.H. 1996. Selenium in birds. In Environmental contaminants in wildlife: Interpreting tissue concentrations. W.N. Beyer, G.H. Heinz and A.W. Redmon- Norwood (eds.). Setac Special Publication, CRC Press, Boca Raton.

Higgs, D.A., Macdonald, J.S., Levings, C.D., Dosanjh, B.S. 1995. Nutrition and feeding habits of Pacific salmon (*Oncorhynchus* species) in relation to life history stage. In Physiological Ecology of Pacific Salmon. C.Groot, L. Margolis and W.C. Clarke (eds.). UBC Press, Vancouver, Canada.

Hunter, B.A., Johnson, M.S. 1982. Food chain relationships of copper and cadmium in contaminated grassland ecosystems. Oikos 38: 108-117.

Janssens, E., Dauwe, T., Bervoets, L. Eens, M. 2002. Inter- and intraclutch variability in heavy metals in feathers of Great tit nestlings (*Parus major*) along a pollution gradient. Arch Environ Contam Toxicol 43: 323-329.

Kingery, H.E. 1996. American Dipper (*Cinclus mexicanus*). The Birds of North America, no.229 (eds. A. Poole and F. Gill), pp.1-28. The Academy of Natural Sciences, Philadelphia, and American Ornithologists' Union, Washington, D.C.

Larison, J.R. Likens, G.E., Fitzpatrick, J.W., Crock, J.G. 2000. Cadmium toxicity among wildlife in the Colorado Rocky Mountains. Nature 406: 181-183.

Laskey, J.W., Edens, F.W. 1985. Effects of chronic high-level manganese exposure on male behavior in the Japanese Quail (*Coturnix coturnix japonica*). Poultry Sci 64: 579-584.

Lawson, N.M., Mason, R.P. 2001. Concentration of mercury, methylmercury, cadmium, lead, arsenic, and selenium in the rain and stream water of two contrasting watersheds in Western Maryland. Water Research 35: 4039-4052.

Lemly, A.D. 1996. Selenium in aquatic organisms. In Environmental contaminants in wildlife: Interpreting tissue concentrations. W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood (eds.). Setac Special Publication, CRC Press, Boca Raton.

Mayack, L.A., Bush, P.B., Fletcher, O.J., Page, R.K. and Fendley, T.T. 1981. Tissue residues of detary cadmium in Wood ducks. Arch Enviorn Contam Toxicol 10: 637-645.

McDaniel, W. 1991. Sample preparation procedure for spectrochemical determination of total recoverable elements in biological tissues. Environmental Services Division, U.S. Environmental Protection Agency. Cincinnati, Ohio, 45268.

Miles, A.K., Grue, C.E. Pendleton, G.W., Soares, J.H. Jr. 1993. Effects of dietary aluminum, calcium and phosphorus on egg and bone of European starlings. Arch Environ Contam Toxicol 24: 206-212.

Miller, C.V., Foster, G.D., Majedi, B.F. 2003. Baseflow and stormflow metal fluxes from two small agricultural catchments in the Coastal Plain of the Chesapeake Bay Basin, United States. Appl Geochem 18: 483-501.

Morrissey, C.A., L.I. Bendell-Young and J.E. Elliott. (submitted-a). Seasonal trends in population density, distribution and movement of American dippers within a watershed of southwestern British Columbia, Canada. Condor.

Morrissey, C.A., Bendell-Young, L.I., Elliott, J.E. (submitted-b). Linking contaminant profiles to the diet and breeding location of American dippers using stable isotope analysis. J Appl Ecol.

Nagy, KA. 1987. Field metabolic rate and food requirement scaling in mammals and birds. Ecol Monogr 57: 111-128.

Nehring, B.R. 1976. Aquatic insects as biological monitors of heavy metal pollution. Bull Environ Contam Toxicol 15: 147-154.

Neugebauer, E.A., Sans Cartier, G.L., Wakeford, B.J. 2000. Methods for the determination of metals in wildlife tissues using various atomic absorption spectrophotometry techniques. Technical Report Series No. 337. Canadian Wildlife Service Headquarters, Hull, Quebec, Canada.

Nimmo, D.R. Willox, M.J., Lafrancois, T.D., Chapman, P.L., Brinkman, S.F., Greene, J.C. 1998. Effects of metal mining and milling on boundary waters of Yellowstone National Park, USA. Environ Manage 22: 913-926.

Nyholm, N.E.I., Myhrberg, H.E. 1977. Severe eggshell defects and impaired reproductive capacity in small passerines in Swedish Lapland. Oikos 29: 336-341.

Nyholm, N.E.I. 1981. Evidence of involvement of aluminum in causation of defective formation of eggshells and of impaired breeding in wild passerine birds. Environ Res 26: 363-371.

Ohlendorf, H., Hoterm, R.L., Walsh, D. 1989. Nest success, cause-specific nest failures and hatchability of aquatic birds at selenium contaminated Kesterson Reservoir and a reference site. Condor 91: 787-796.

Ohlendorf, H., Kilness, A.W., Simmons, J.L., Stroud, R.K., Hoffman, D.J., Moore, J.F. 1988. Selenium toxicosis in wild aquatic birds. J Toxicol Environ Health 24: 67-92.

Ormerod, S.J., Bull, K. R., Cummins, C.P., Tyler, S.J., Vickery, J.A. 1988. Egg mass and shell thickness in dippers *Cinclus cinclus* in relation to stream acidity in Wales and Scotland. Environ Pollut 55: 107-121.

Sadler, K. Lynam, S.S. 1985. The mineral content of some freshwater invertebrates in relation to stream pH and calcium concentration. Central Electricity Generating Board Report. TPRD/L/2781/N84.

Sample B.E., Opresko D.M., Suter G.W. II. 1996. Toxicological benchmarks for wildlife. 1996 Revision. ES/ER/TM-86/R3. Oak Ridge National Laboratory, Oak Ridge, TN, USA.

Scheuhammer, A.M. 1987. The chronic toxicity of aluminum, cadmium, mercury, and lead in birds: a review. Environ Pollut 46: 263-295.

Scheuhammer, A.M. 1991. Effects of acidification on the availability of toxic metals and calcium to wild birds and mammals. Environ Pollut 71: 329-375.

Schindler, D.W., Kaslan, S.E.M., Hesslein, R.H. 1989. Estimating biological damage to lakes of the Midwestern and northeastern United states from acid rain. Environ Sci Technol 23: 573-580.

Scott, N.A., Likens, G.E., Eaton, J.S., Siccama, T.G. 2001. Trace metal loss following whole-tree harvest of northeastern deciduous forest, USA. Biogeochem 54: 197-217.

Spahn, S.A., Sherry, T.W. 1999. Cadmium and lead exposure associated with reduced growth rates, poorer fledging success of Little Blue heron chicks (*Egretta caerulea*) in South Louisiana wetlands. Arch Environ Contam Toxicol 37: 377-384.

Sparling, D.W. 1990. Acid precipitation and food quality: Inhibition of growth and survival in black ducks and mallards by dietary aluminum, calcium, and phosphorus. Arch Environ Contam Toxicol 19: 457-463.

Strom, S.M., Ramsdell, H.S., Archuleta, A.S. 2002. Aminolevulinic acid dehydratase ctivity in American dippers (*Cinclus mexicanus*) from a metal-impacted stream. Environ Toxicol and Chem 21: 115-120.

Swain, L.G. 1987. Second report on chemical sensitivity of B.C. lakes to acidic inputs. Water Management Branch, Environment and Resources Division, Ministry of Water, Land and Air Protection

Wiener, J.G., Spry, D.J. 1996. Toxicological significance of mercury in freshwater fish. In Environmental contaminants in wildlife: Interpreting tissue concentrations. W.N. Beyer, G.H. Heinz and A.W. Redmon- Norwood (eds.). Setac Special Publication, CRC Press, Boca Raton.

FIGURE LEGENDS

Figure 6.1 a-d. Calculated daily exposure of a) cadmium b) copper c) lead and d) zinc (μg/g body weight per day) to American dippers breeding on the main river or on the tributaries in the Chilliwack watershed with diets of increasing proportions of fish relative to invertebrates (i.e. 25 % fish = 75 % invertebrates). Data from invertebrates and fish are pooled for 2000 and 2001. Tolerable daily intake (TDI) levels (μg/g body weight per day) are shown for assessment of exposure risks except Pb, which was well below TDI.

Figure 6.2 a-e. Calculated daily exposure of a) selenium b) aluminum c) arsenic d) manganese and e) mercury (μg/g body weight per day) to American dippers breeding on the main river or on the tributaries in the Chilliwack watershed with diets of increasing proportions of fish relative to invertebrates (i.e. 25 % fish = 75 % invertebrates). Data from invertebrates and fish are pooled for 2000 and 2001. Tolerable daily intake (TDI) levels (μg/g body weight per day) are shown for assessment of exposure risks except Mn, which was well below TDI.

Figure 6.1a

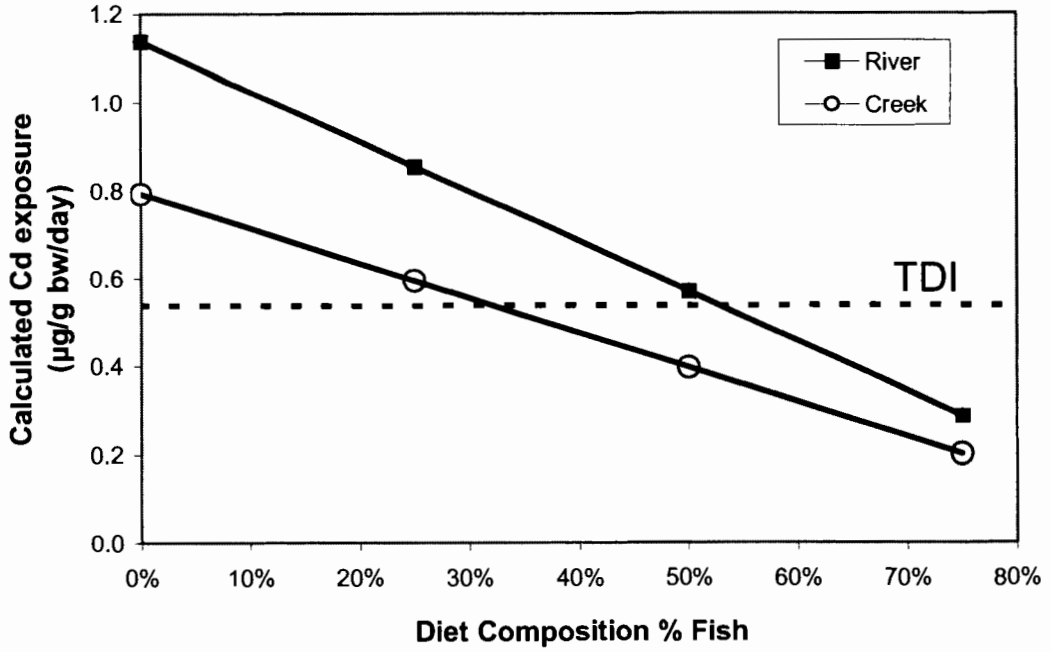
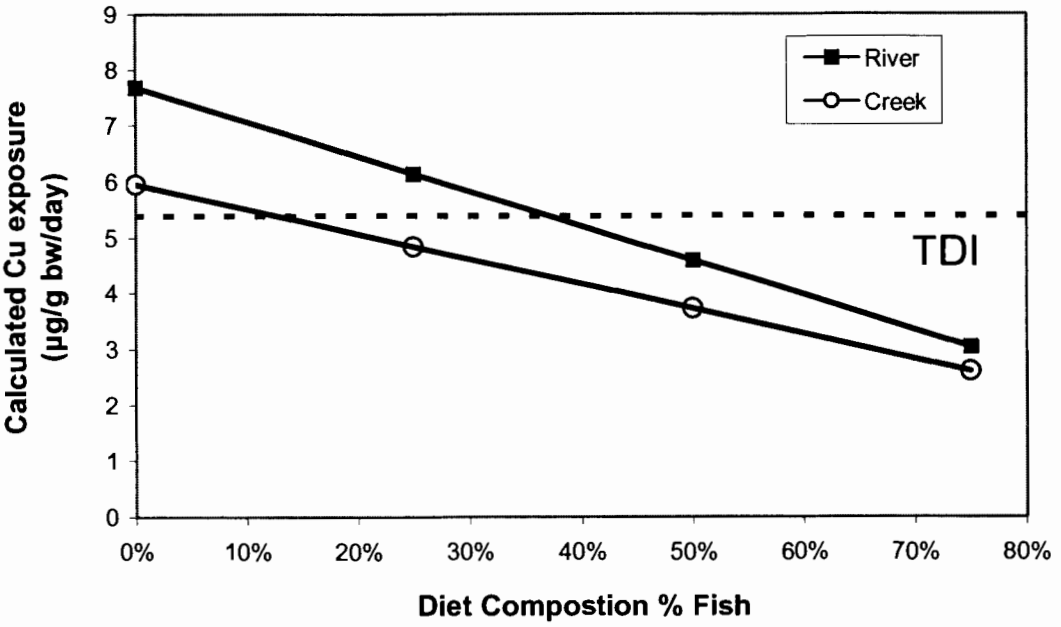


Figure 6.1b



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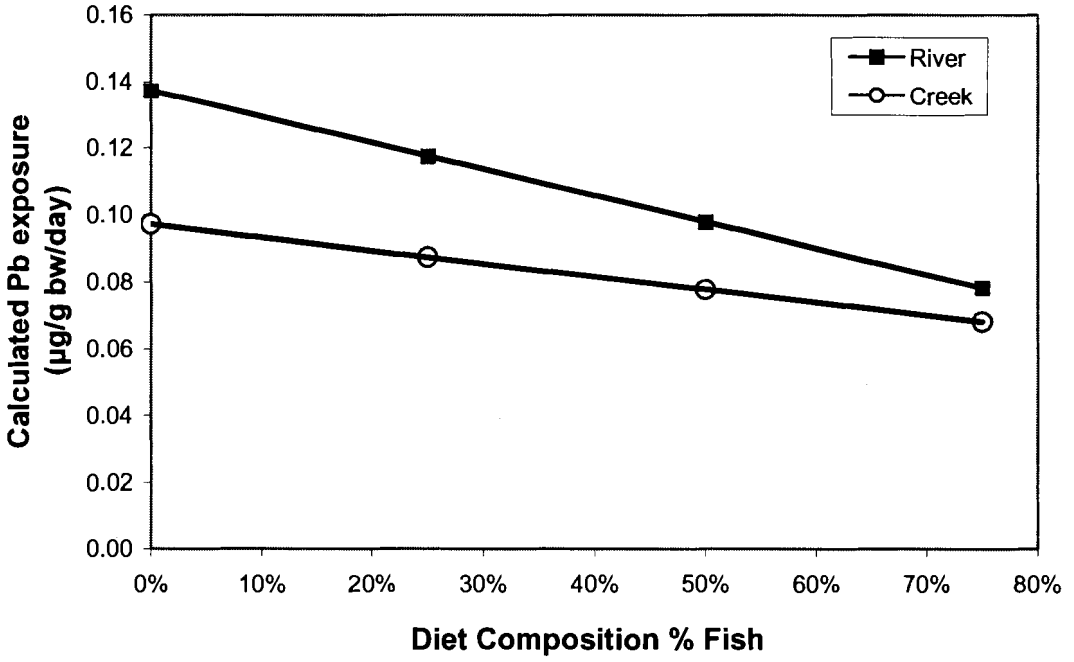


Figure 6.1d

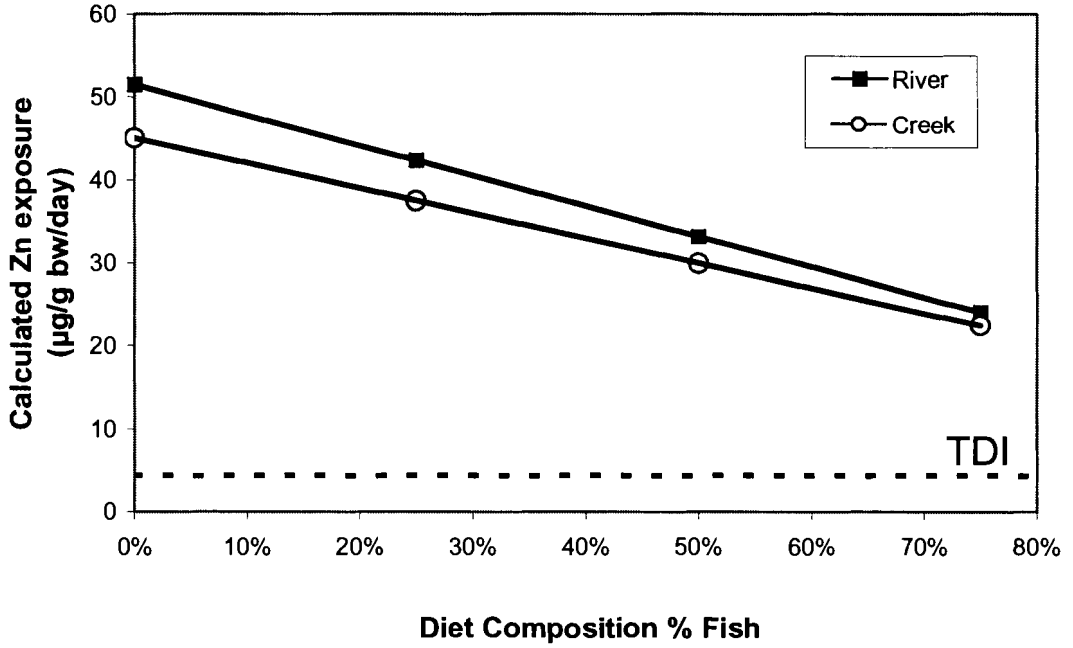


Figure 6.2a

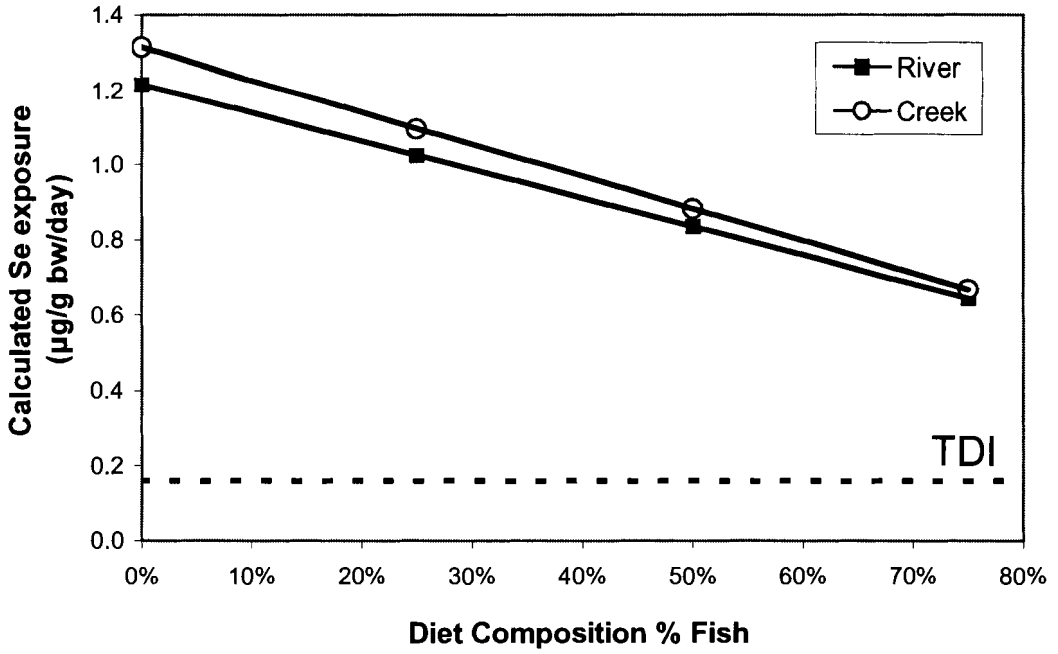
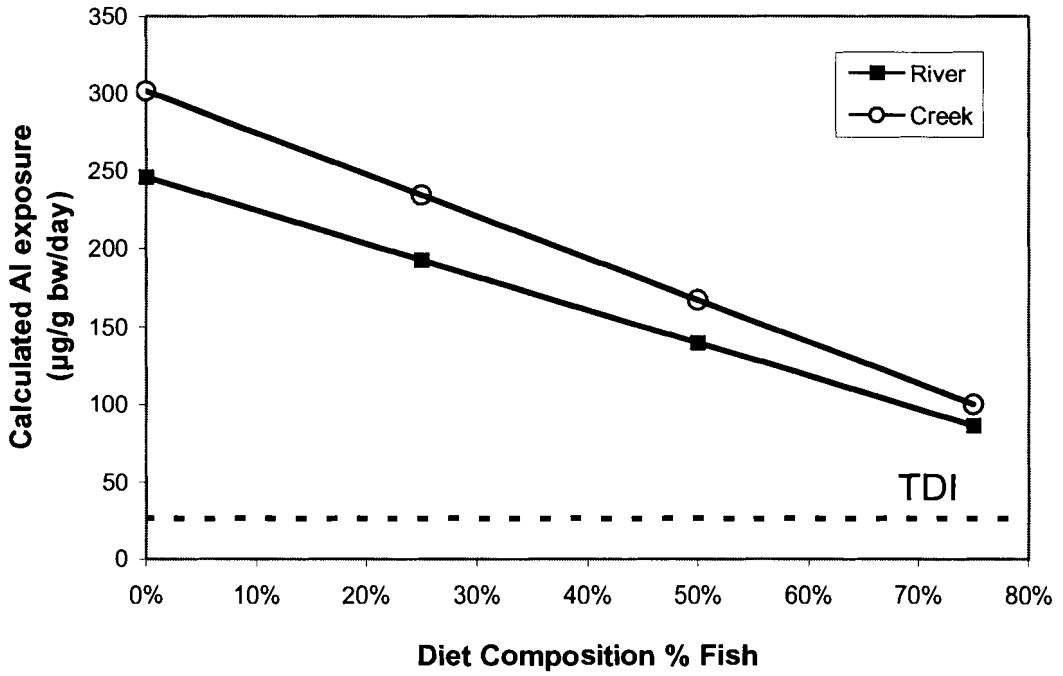


Figure 6.2b



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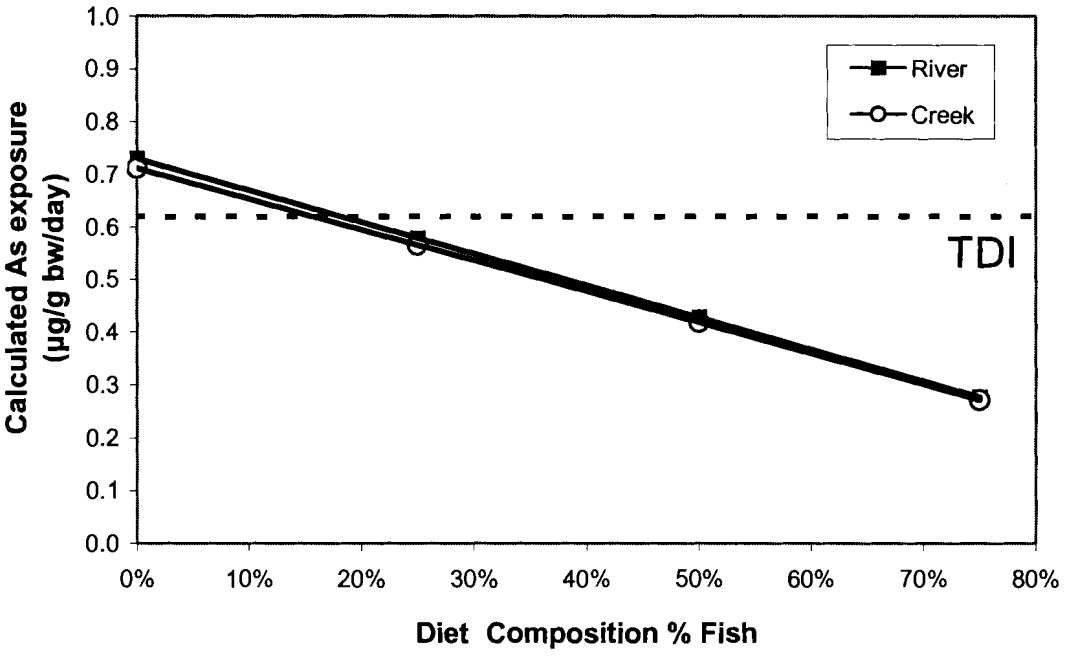

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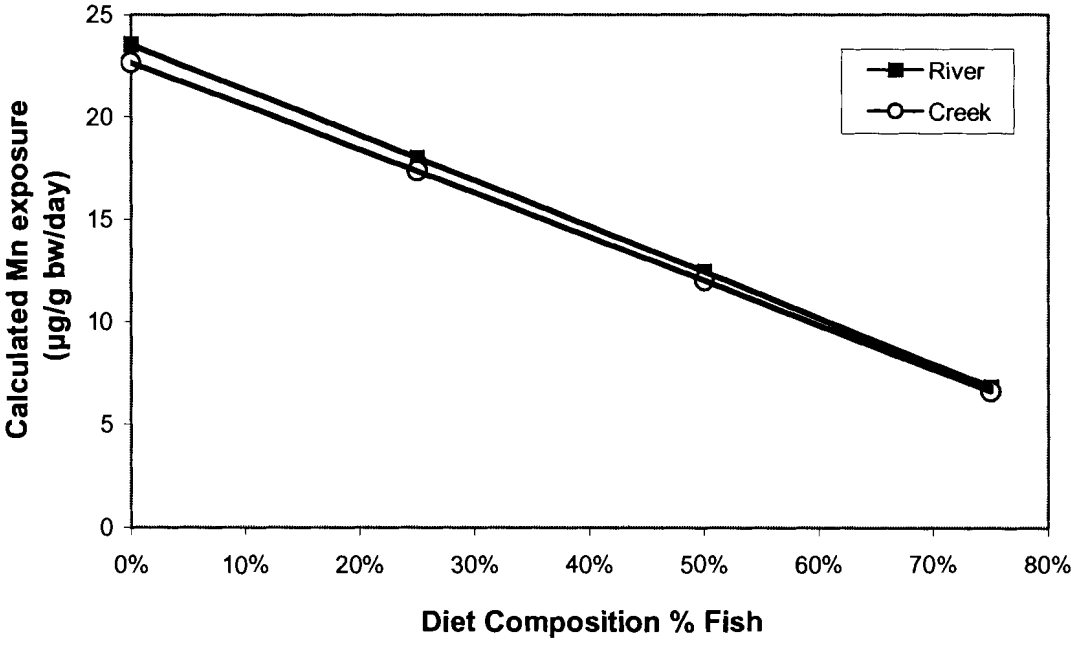

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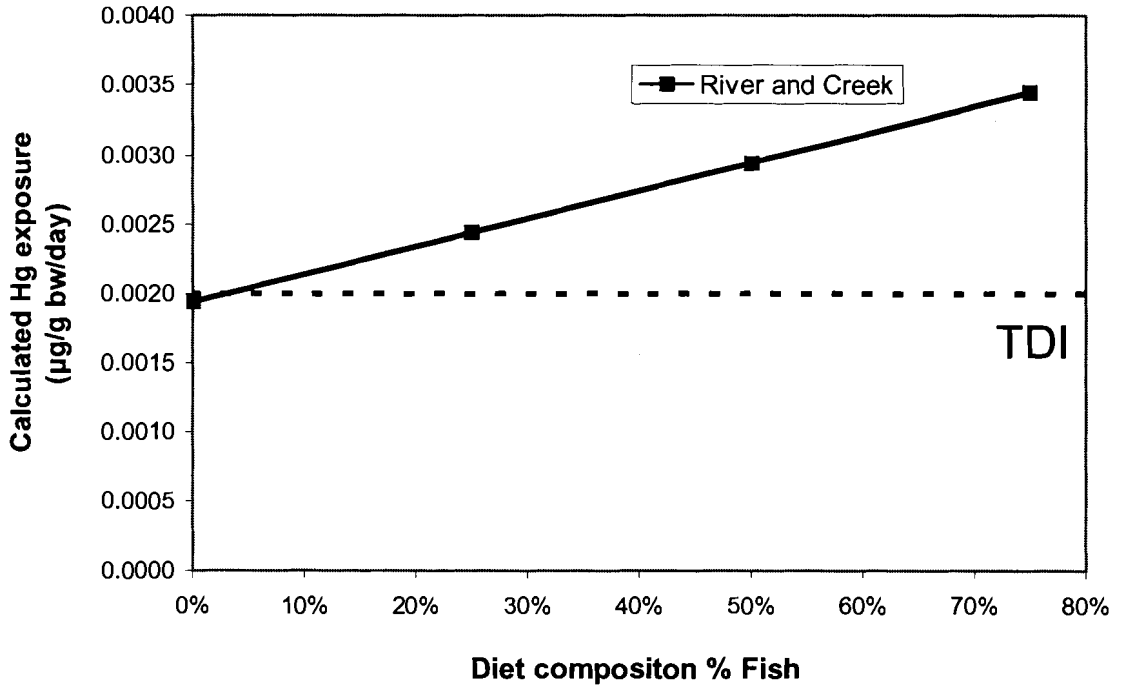
0.0040
0.0035
0.0030
0.0025
0.0020
0.0015
0.0010
0.0005
0.0000
Calculated Hg exposure (µg/g bw/day)
River and Creek
TDI
0%
10%
20%
30%
40%
50%
60%
70%
80%
Diet compositon % Fish

TABLE 6.1 Summary of trace metal concentrations and frequency of detection for aquatic invertebrates and salmon fry (fish) from the Chilliwack River watershed, British Columbia. Given are arithmetic means (μg/g dry weight) ± SE with geometric means in parentheses below. Data for 2000 and 2001 are combined.

Metal	River Invertebrates	Creek Invertebrates	Fish	*Significance (p)	Invertebrate % Detected	Fish % Detected
Hg	0.005 ± 0.005 (0.005)[a]	0.018 ± 0.004 (0.011)[a]	0.04 ± 0.01 (0.02)[b]	0.002	20	47
Cd	4.58 ± 0.40 (4.31)[a]	3.66 ± 0.40 (3.35)[a]	1.37 ± 0.19 (1.27)[b]	<0.0001	100	100
Pb	0.67 ± 0.12 (0.58)[a]	0.55 ± 0.15 (0.41)[a]	0.42 ± 0.10 (0.33)[a]	NS	70	100
Se	5.83 ± 0.74 (5.55)[a]	6.08 ± 0.79 (5.14)[a]	2.68 ± 0.27 (2.58)[b]	0.006	100	100
Cu	33.29 ± 1.96 (32.48)[a]	26.43 ± 2.10 (25.17)[b]	9.05 ± 1.23 (8.39)[c]	<0.0001	100	100
Mn	129.47 ± 29.65 (99.77)[a]	107.90 ± 17.4 (96.08)[a]	8.56 ± 2.13 (7.07)[a]	<0.0001	100	100
Zn	228.60 ± 17.10 (217.63)[a]	203.31 ± 18.91 (190.47)[a]	87.76 ± 7.84 (84.89)[b]	<0.0001	100	100
Al	1296.44 ± 215.98 (1040.32)[a]	1585.99 ± 303.72 (1275.76)[a]	165.53 ± 49.38 (119.91)[b]	<0.0001	100	100
As	3.73 ± 0.50 (3.09)[a]	3.77 ± 0.65 (3.01)[a]	0.63 ± 0.10 (0.56)[b]	<0.0001	100	100

*Geometric means with the same letter are not significantly different using one-way ANOVA and Tukey multiple comparison procedure ($\alpha = 0.05$).

TABLE 6.2. Summary of trace metal concentrations and frequency of detection in feathers of adult resident and migrant American dippers from the Chilliwack River watershed, British Columbia, 1999-2001. Given are arithmetic means (μg/g dry weight) ± SE with geometric means in parentheses below.

Metal	N	River Resident Mean ● SE	River Resident % Detected	Creek Migrant Mean ● SE	Creek Migrant % Detected	Significance (p)
Hg	105	0.79 ± 0.06 (0.65)	96.4	0.58 ± 0.06 (0.50)	98.0	0.04
Cd	39	0.25 ± 0.04 (0.19)	45.0	0.13 ± 0.04 (0.12)	52.5	0.01
Pb	72	1.06 ± 0.21 (0.59)	92.5	0.89 ± 0.21 (0.57)	87.5	ns[a]
Se	73	6.01± 0.32 (5.51)	97.5	5.82 ± 0.36 (5.75)	85.0	ns
Cu	79	14.32 ± 2.08 (10.65)	100.0	9.65 ± 2.14 (7.29)	97.5	0.05
Mn	73	1.4 ± 0.21 (0.71)	87.5	0.78 ± 0.21 (0.63)	92.5	ns
Zn	80	129.73 ± 3.45 (127.59)	100.0	131.87 ± 3.54 (130.57)	100.0	ns
Al	13	48.32 ± 34.1 (20.08)	12.5	59.03 ± 26.96 (23.94)	20.0	ns
As	1	12.28	2.5	ND[b]	ND	ns

[a]ns = Not significant ($p > 0.05$). [b]ND = non detectable in sample.

TABLE 6.3. Summary of arithmetic (geometric) mean concentrations (ug/g dry weight) ± SE and frequency of detection in adult and nestling fecal samples collected from the Chilliwack River watershed, British Columbia in 2001.

Metal	Feces Mean ± SE	% Detected (n = 14)
Hg	0.036 ± 0.005 (0.031)	93
Cd	5.97 ± 1.04 (4.89)	100
Pb	3.65 ± 1.03 (2.50)	100
Se	4.83 ± 0.43 (4.55)	100
Cu	53.28 ± 4.54 (50.58)	100
Mn	311.53 ± 47.28 (259.42)	100
Zn	396.10 ± 42.09 (370.68)	100
Al	2780.5 ± 388.30 (2312.07)	100
As	5.05 ± 1.03 (4.11)	100

APPENDICES

Appendix 1. Summary of toxicity data used to calculate Tolerable Daily Intake (TDI) for American dippers. Selected toxicity tests on suitable avian species to obtain a NOAEL and LOAEL are adapted from Sample et al. 1996. Means of two or more suitable toxicity tests are used where applicable. For calculating TDI for Hg, a toxicity test for MeHg was used. See Sample et al. (1996) for full description of methods and references.

Appendix 2. Concentrations (mg/kg) of trace metals analyzed by ICP-AES in benthic invertebrates from the Chilliwack River watershed, British Columbia. Samples were collected from eight sites on the main river and seven different tributaries in 2001.

Appendix 3. Concentrations (mg/kg) of trace metals analyzed by ICP-MS in American Dipper fecal samples collected from adults and nestlings in the Chilliwack River watershed, 2001.

Appendix 4. Concentrations (mg/kg) of trace metals analyzed by ICP-MS in feather samples collected from resident and migrant adult American Dippers in the Chilliwack River watershed, 1999-2001. Feathers collected from adults and juveniles in 1999 were analyzed for Hg only and are shown in the last table.

Appendix 1

Metal	Metal Form	Test Species	NOAEL (mg/kg/d)	LOAEL (mg/kg/d)	Uncertainty factor	TDI	Reference
Al	$Al_2(So_4)_3$	ringed dove	109.7	614.3	10	**25.96**	Carriere et al. 1986
As	sodium arsenite	mallard	5.14	12.84	10	0.81	USFWS 1964
	paris green	brown-headed cowbird	2.46	7.38	10	0.43	USFWS 1969
		AVERAGE	3.8	10.11	10	**0.62**	
Cd	cadmium chloride	mallard	1.45	20.03	10	**0.54**	White and Finley 1978
Cu	copper oxide	1 day old chicks	47	61.7	10	**5.39**	Mehring et al.1960
Pb	lead acetate	Japanese quail	1.13	11.3	10	0.36	Edens et al. 1976
	metallic	American kestrel	3.85	21.56	10	0.91	Pattee 1984
		AVERAGE	2.49	16.43	10	**0.64**	
Hg	methyl mercury	mallard	0.0064	0.064	10	**0.002**	Heinz 1979
	mercuric chloride	Japanese quail	0.45	0.9	10	0.064	Hill and Schaffner 1976
Mn	Mn_3O_4	Japanese quail	997	5583	10	**235.9**	Laskey and Edens 1985
Se	sodium selenite	mallard	0.5	1	10	0.071	Heinz et al. 1987
	selenomethionine	mallard	0.4	0.8	10	0.057	Heinz et al. 1989
	selenomethionine	screech owl	0.44	1.5	10	0.081	Wiemeyer and Hoffman 1996
	selenomethionine	black crowned night heron	1.8	10.08	10	0.426	Smith et al. 1988
		AVERAGE	0.785	3.345	10	**0.162**	
Zn	zinc sulfate	white leghorn chicken	14.5	131	10	**4.36**	Stahl et al. 1990

Appendix 2

Invertebrate ICP-AES analysis 2001: Concentrations in mg/kg dry weight (ppm)												
Chilliwack River Invertebrates (all metals)												
Sample	**Location**	**Type**	**Date**	**Sample dry**	**Ag**	**Al**	**As**	**B**	**Ba**	**Be**	**Bi**	**Ca**
			Collected	**Weight (g)**								
***Detection Limit (ppm)**					**0.05**	**2.41**	**0.24**	**2.41**	**0.10**	**0.05**	**2.41**	**4.82**
1	Vedder	invertebrate	26-Apr-01	1.085	0.056	1636.43	6.11	5.06	41.97	0.054	ND	5744.02
2	Edwards	invertebrate	26-Apr-01	1.200	ND	931.07	5.48	2.78	16.87	ND	ND	3413.65
3	Chwk R/Tamihi	invertebrate	24-Apr-01	1.013	0.082	1248.85	6.04	2.47	25.50	ND	ND	3359.90
4	Anderson	invertebrate	24-Apr-01	1.199	0.068	382.12	4.67	2.88	16.96	ND	ND	3906.58
5	Thurston	invertebrate	24-Apr-01	1.116	0.109	785.85	5.24	3.21	17.60	ND	ND	3614.54
6	Hatchery (wild)	invertebrate	25-Apr-01	1.070	0.085	484.94	4.07	3.23	37.90	ND	ND	2914.80
7	Camp Foley	invertebrate	23-Apr-01	0.999	0.166	373.87	4.75	ND	55.68	ND	ND	2256.82
8	Riverside	invertebrate	26-Apr-01	0.986	0.236	2225.70	7.08	3.75	150.62	0.057	ND	3995.53
9	Tamihi Creek	invertebrate	24-Apr-01	1.083	ND	625.48	4.99	6.61	17.76	ND	ND	2506.88
10	Slesse Creek	invertebrate	25-Apr-01	0.957	0.138	537.70	4.76	4.98	117.46	ND	ND	5318.22
11	Nesackwatch Creek	invertebrate	1-May-01	0.957	0.157	663.50	4.76	4.24	32.42	ND	ND	2713.30
12	Foley Creek	invertebrate	23-Apr-01	0.917	0.134	821.55	5.44	ND	93.66	ND	ND	3129.17
13	Liumchen Creek	invertebrate	25-Apr-01	0.977	0.069	2141.13	6.29	4.69	34.22	0.054	ND	7374.83
14	Borden Creek	invertebrate	24-Apr-01	1.049	0.075	1013.00	5.78	2.72	24.82	ND	ND	8497.57
15	Chipmunk Creek	invertebrate	30-Apr-01	0.947	0.426	4319.53	8.95	7.97	65.95	0.091	ND	3499.38

*Detection Limit (DL) in mg/kg dw (ppm) based on average sample weight of 1.037g.

Appendix 2

Invertebrate ICP-AES analysis 2001: Concentrations in mg/kg dry weight (ppm)														
Chilliwack River Invertebrates (all metals)														
Sample	**Cd**	**Co**	**Cr**	**Cu**	**Fe**	**Hg**	**K**	**La**	**Mg**	**Mn**	**Mo**	**Na**	**Ni**	**P**
***DL (ppm)**	**0.05**	**0.24**	**0.24**	**0.10**	**0.48**	**0.02**	**48.22**	**0.24**	**4.82**	**0.48**	**0.10**	**48.22**	**0.24**	**0.48**
1	2.37	3.51	3.54	33.62	2481.65	ND	7714.3	0.000	2145.5	126.56	1.32	3645	3.222	13218.56
2	3.85	5.56	1.89	33.00	1625.89	ND	7921.9	0.000	1516.3	108.73	1.06	3576	2.420	12407.66
3	5.94	6.74	2.90	36.38	2253.54	ND	8057.1	0.000	1731.5	111.40	1.08	3299	3.366	12357.31
4	6.36	4.44	0.62	34.39	697.06	ND	8765.8	0.000	1696.6	78.32	1.05	4095	2.505	15031.62
5	6.48	3.52	2.23	30.39	1561.65	ND	8852.3	0.000	1780.7	64.34	1.10	3961	2.489	14462.43
6	5.33	2.98	0.75	28.85	958.98	ND	8641.1	0.000	1670.7	64.11	0.91	4133	2.034	14268.53
7	7.71	4.14	0.50	33.68	642.70	ND	9097.9	0.000	1396.9	49.28	1.11	4328	1.516	13341.58
8	5.58	4.48	3.13	40.78	3797.10	ND	10372.5	0.000	2515.4	106.77	1.57	4431	2.305	14463.11
9	6.34	6.60	1.08	31.61	976.24	ND	8238.8	0.000	1402.3	57.94	0.93	3926	2.124	12216.09
10	2.99	2.26	1.07	36.04	1012.12	ND	9492.3	0.000	2144.5	50.58	0.95	4359	1.381	17994.82
11	1.39	2.91	1.41	23.03	1519.11	ND	9471.2	0.000	1850.0	93.90	1.49	3831	3.257	14366.58
12	3.97	4.12	1.60	29.84	1672.94	0.04	10192.1	0.000	1594.1	109.15	1.14	4559	3.274	14706.26
13	3.44	5.31	2.50	34.88	4426.77	0.02	8351.7	0.000	2184.3	111.81	1.53	4549	7.278	15596.54
14	4.63	3.11	1.18	24.20	1399.21	ND	10185.7	0.000	1863.9	68.95	0.82	4236	1.593	14930.72
15	5.48	4.43	3.08	30.70	6804.10	0.04	10292.4	0.000	2215.9	312.28	2.00	3684	3.611	12365.03

*Detection Limit (DL) in mg/kg dw (ppm) based on average sample weight of 1.037g.

Appendix 2

Invertebrate ICP-AES analysis 2001: Concentrations in mg/kg dry weight (ppm)										
Chilliwack River Invertebrates (all metals)										
Sample	**Pb**	**S**	**Sb**	**Se**	**Si**	**Sn**	**Sr**	**Ti**	**V**	**Zn**
***DL (ppm)**	**0.48**	**48.22**	**1.45**	**0.24**	**4.82**	**0.48**	**0.10**	**0.24**	**0.24**	**0.48**
1	0.84	5627.08	ND	6.42	104.9	2.95	15.38	27.38	5.19	229.66
2	ND	6263.21	ND	7.13	47.3	2.16	9.66	29.52	3.20	227.98
3	0.69	6453.24	ND	6.91	41.0	2.73	11.86	19.23	5.05	264.47
4	0.67	5843.84	ND	7.00	36.5	2.04	10.20	17.89	1.34	333.58
5	0.56	6511.51	ND	7.23	36.0	2.57	10.22	29.75	2.97	267.16
6	0.49	5880.79	ND	5.67	19.6	1.78	8.52	20.19	1.91	278.21
7	ND	7055.50	ND	3.88	32.2	1.56	7.34	18.00	1.22	260.67
8	ND	7201.64	ND	8.48	86.5	3.10	13.38	19.54	10.05	297.98
9	ND	6647.51	ND	7.87	28.9	1.66	8.66	28.33	2.22	213.36
10	ND	5384.26	ND	4.87	48.1	2.67	12.86	12.78	2.35	276.60
11	ND	6745.97	ND	5.87	25.1	2.91	8.74	41.20	3.61	235.08
12	ND	7212.22	ND	6.28	27.4	2.22	15.75	31.67	2.73	316.91
13	ND	6225.89	ND	9.09	63.2	3.85	29.97	64.46	9.38	276.70
14	ND	7331.49	ND	8.02	89.5	2.44	15.11	33.84	2.40	214.24
15	1.52	6995.07	ND	17.40	49.5	3.84	22.85	26.30	10.78	221.74

*Detection Limit (DL) in mg/kg dw (ppm) based on average sample weight of 1.037g.

Appendix 3

Fecal ICP-MS analysis: Concentrations in mg/kg dry weight (ppm)												
American dipper fecal samples (all metals)												
Sample	**Sample Name**	**Type**	**Date**	**Sample dry**	**Ag**	**Al**	**As**	**B**	**Ba**	**Be**	**Bi**	**Ca**
			Collected	**Weight (g)**								
***Detection Limit (ppm)**					**0.004**	**0.04**	**0.13**	**0.42**	**0.004**	**0.008**	**0.004**	**2.08**
1	Anderson #5	nestling fecal	23-Jul-01	0.419	0.151	4717.83	8.112	3.70	228.48	0.117	0.068	2423.66
2	CHK99-014	adult fecal	23-Jul-01	0.341	0.103	4167.80	4.817	4.40	151.62	0.100	0.043	3028.68
3	Lower Hatchery	nestling fecal	7-May-01	0.439	0.151	508.22	1.214	3.87	14.93	0.007	0.007	3638.34
4	CHK99-011	adult fecal	5-Jun-01	0.478	0.130	1930.33	5.273	5.02	26.82	0.021	0.011	6496.95
5	Riverside Rec	nestling fecal	26-Apr-01	0.224	0.124	599.73	1.511	8.48	87.06	0.027	0.008	28380.43
6	CHK99-202	adult fecal	28-Jun-01	0.099	0.082	5353.84	3.295	24.24	96.65	0.133	0.048	21013.32
7	CHK00-316	adult fecal	12-Jun-01	0.127	0.170	2549.13	4.795	14.17	77.21	0.063	0.032	9796.99
8	Borden Creek	nestling fecal	13-Jun-01	0.284	0.205	2542.01	3.115	7.04	101.61	0.067	0.021	23237.54
9	CHK99-111	adult fecal	5-Jun-01	0.236	0.110	4080.42	5.600	8.47	67.03	0.076	0.045	6029.74
10	CHK00-364	adult fecal	2-May-01	0.121	0.046	2802.07	3.133	12.40	39.54	0.041	0.030	9352.18
11	Lower Tamihi #3	nestling fecal	20-Apr-01	0.179	0.058	1279.72	2.776	15.64	23.36	0.034	0.022	3011.59
12	CHK00-336	adult fecal	2-May-01	0.158	0.059	2336.77	4.842	10.13	32.98	0.063	0.014	5747.80
13	CHK99-117	adult fecal	2-May-01	0.046	ND	3283.04	5.311	28.26	90.05	0.065	0.036	13779.87
14	CHK00-339	adult fecal	23-Jul-01	0.206	0.082	2775.53	16.950	10.19	97.40	0.068	0.042	6293.49

*Detection Limit (DL) in mg/kg (ppm) based on avg. sample weight of 0.240g.

Appendix 3

Fecal ICP-MS analysis: Concentrations in mg/kg dry weight (ppm)														
American dipper fecal samples (all metals)														
Sample	**Cd**	**Co**	**Cu**	**Cr**	**Fe**	**Hg**	**K**	**Li**	**Mg**	**Mn**	**Mo**	**Na**	**Ni**	**P**
***DL (ppm)**	**0.004**	**0.004**	**0.004**	**0.04**	**0.42**	**0.002**	**2.08**	**0.042**	**0.42**	**0.004**	**0.004**	**0.42**	**0.08**	**0.83**
1	5.2134	9.2499	51.97	10.290	8137.17	0.029	10510.70	6.134	4610.86	562.06	2.142	3037.23	13.415	6992.76
2	3.1212	7.5063	48.92	10.400	7911.71	0.006	6797.58	5.073	3489.15	484.37	1.487	1421.99	11.383	4686.34
3	1.7681	1.3278	79.17	1.366	1028.70	0.020	15753.50	0.569	4272.89	84.36	0.878	6908.88	1.548	17198.90
4	11.5338	11.7099	55.54	3.724	3480.31	0.048	12974.76	1.904	2728.66	135.96	1.120	2773.01	6.069	12905.65
5	3.3603	1.5823	19.74	1.419	2212.61	0.041	16713.92	0.759	3652.68	102.88	1.181	5579.46	1.832	24981.17
6	1.4731	5.1721	42.68	12.367	10345.66	0.049	10298.07	7.374	4103.03	221.45	2.060	1880.81	11.491	11181.06
7	6.2050	7.8050	92.57	5.548	5951.24	0.055	10347.17	3.228	3786.61	520.91	2.069	4188.19	8.377	10325.22
8	4.4793	6.1663	51.22	3.238	4509.87	0.054	11984.14	3.556	4880.28	425.25	0.863	3222.18	4.459	19242.18
9	14.6092	13.4110	62.22	8.814	7611.42	0.051	12169.28	4.280	4128.39	442.11	1.275	2331.78	10.594	12622.79
10	3.7551	5.0956	53.44	8.107	6914.77	0.021	14749.32	3.140	3975.21	195.15	0.962	3334.71	7.614	13204.53
11	8.3618	7.3217	42.35	5.255	2411.73	0.037	11350.47	1.564	1775.98	136.48	1.200	4867.04	5.203	6954.90
12	10.6472	11.5216	47.69	4.179	4787.16	0.046	12831.63	2.722	2932.91	188.87	1.052	3531.01	7.312	12426.16
13	4.7561	8.1622	46.80	8.816	7019.79	-0.015	12679.48	3.478	4539.13	527.33	1.823	4160.87	11.843	16279.23
14	4.2658	10.4452	51.55	10.271	7273.08	0.013	12289.25	3.107	4006.80	334.32	2.415	4133.01	12.596	9607.60

*Detection Limit (DL) in mg/kg (ppm) based on avg. sample weight of 0.240g.

Appendix 3

Fecal ICP-MS analysis: Concentrations in mg/kg dry weight (ppm)												
American dipper fecal samples (all metals)												
Sample	**Pb**	**Sb**	**Se**	**Sn**	**Sr**	**Te**	**Ti**	**Tl**	**U**	**V**	**Zn**	**Zr**
***DL (ppm)**	**0.004**	**0.004**	**0.125**	**0.004**	**0.004**	**0.008**	**0.042**	**0.004**	**0.004**	**0.42**	**0.04**	**0.004**
1	3.28	0.027	3.129	0.0297	16.23	0.008	84.47	0.058	4.702	21.56	392.23	0.832
2	2.57	0.014	1.932	0.0133	14.47	0.028	60.38	0.078	1.746	32.37	203.28	0.589
3	0.47	0.027	4.099	0.0070	26.91	0.005	30.58	0.027	0.311	2.43	245.39	0.098
4	1.05	0.034	6.441	0.0310	26.29	0.015	103.56	0.037	0.412	8.79	319.75	0.763
5	0.95	0.024	4.713	0.0327	65.37	ND	76.98	0.040	2.408	4.25	374.89	0.085
6	4.00	0.050	3.297	0.0369	52.79	0.035	501.94	0.061	2.623	26.44	417.14	0.962
7	2.82	0.057	5.521	ND	49.94	0.025	210.89	0.045	2.919	12.43	745.15	0.252
8	1.68	0.039	5.551	0.0482	57.47	0.025	102.29	0.022	0.263	7.27	712.72	0.868
9	2.71	0.029	4.914	0.0365	26.41	0.020	109.86	0.061	0.858	19.61	438.83	1.291
10	5.25	0.030	5.463	ND	37.36	0.026	237.14	0.056	0.287	18.23	266.12	1.020
11	15.49	0.085	5.446	0.0236	22.85	0.012	123.79	0.049	0.164	6.04	286.43	0.502
12	1.28	0.049	8.380	0.0334	25.34	0.029	168.65	0.051	0.224	10.08	342.30	0.860
13	6.95	0.217	5.477	ND	42.75	0.044	283.58	0.049	0.564	17.92	402.69	1.406
14	2.52	0.040	3.202	0.0352	22.15	0.023	217.34	0.043	3.521	21.61	398.51	1.599

*Detection Limit (DL) in mg/kg (ppm) based on avg. sample weight of 0.240g.

Appendix 4

Feather ICP-MS analysis: Concentrations in mg/kg dry weight (ppm)

Chilliwack watershed American Dipper feather samples

Sample	Bird/Sample ID	Group	Sex	Age	Date Collected	Location	Sample dry Weight (g)	Ag	Al	As	B	Ba	Be	Bi	Ca	Cd	Co	Cu	Cr
***Detection limit (ppm)**								**0.08**	**0.77**	**2.31**	**0.08**	**0.08**	**0.15**	**0.08**	**38.46**	**0.08**	**0.08**	**0.08**	**0.77**
1	CHK99-009b	migrant	Female	Adult	28-Feb-01	Camp Foley	0.020	0.411	ND	ND	ND	2.788	ND	ND	3061.06	0.069	0.159	10.94	ND
2	CHK99-011b	resident	Female	Adult	22-May-01	Tamihi	0.015	0.328	ND	ND	ND	7.890	ND	ND	3671.13	0.979	0.342	6.84	0.493
3	CHK99-014b	resident	Female	Adult	12-Jul-01	Anderson 5b	0.018	0.329	ND	ND	21.07	11.553	ND	ND	3316.42	0.220	0.361	9.14	ND
4	CHK99-111b	resident	Male	Adult	24-May-01	Tamihi	0.017	0.175	ND	ND	78.45	9.059	ND	0.077	3946.97	0.064	0.132	20.76	0.291
5	CHK99-126	migrant	Unknown	Adult	3-Mar-99	Thurston	0.006	0.420	9.287	ND	75.34	11.665	ND	ND	4108.09	0.198	0.278	19.54	ND
6	CHK99-132	migrant	Male	Adult	4-Mar-99	Camp Foley	0.008	0.251	20.269	ND	99.63	0.415	ND	0.213	2809.74	ND	0.300	7.64	ND
7	CHK99-156b	migrant	Female	Adult	21-Feb-01	Slesse Creek	0.014	0.176	ND	ND	108.26	11.196	ND	0.066	3619.31	0.154	0.102	6.78	ND
8	CHK99-186	migrant	Female	Adult	13-May-99	Foley Creek	0.009	0.189	ND	ND	76.98	4.647	ND	ND	2514.59	ND	0.209	1.70	0.618
9	CHK99-187	migrant	Male	Adult	13-May-99	Foley Creek	0.010	0.110	5.190	ND	50.61	5.917	ND	ND	2564.65	ND	0.115	2.89	0.684
10	CHK99-193b	resident	Male	Adult	2-Mar-00	Thurston	0.010	0.114	27.818	ND	171.95	7.977	ND	0.021	2740.95	ND	ND	5.02	1.409
16	CHK00-015	migrant	Female	Adult	19-Jan-00	Hatchery	0.012	0.109	ND	ND	21.72	5.885	ND	0.017	2297.89	ND	0.138	1.13	0.652
17	CHK00-017	migrant	Female	Adult	19-Jan-00	Chwk River/Tamihi	0.012	0.088	16.215	ND	16.69	1.571	ND	ND	2523.53	ND	0.153	0.09	0.539
18	CHK00-019	migrant	Female	Adult	20-Jun-00	Liumchen/Edwards	0.012	ND	ND	ND	108.94	5.367	ND	0.069	2993.22	0.105	0.180	3.64	ND
19	CHK00-021b	migrant	Female	Adult	16-Apr-01	Camp Foley	0.010	ND	16.038	ND	ND	8.657	ND	ND	2699.82	0.119	0.159	ND	0.988
20	CHK00-025	migrant	Female	Adult	8-Feb-00	Thurston	0.009	ND	ND	ND	30.28	12.273	ND	ND	2672.04	ND	0.189	7.20	0.000
21	CHK00-028	migrant	Female	Adult	9-Feb-00	Upper Vedder	0.012	0.112	ND	ND	44.06	6.630	ND	ND	2069.33	0.086	0.105	2.82	0.619
22	CHK00-034	resident	Female	Adult	16-Feb-00	Little Tamihi	0.011	ND	0.562	ND	ND	9.848	ND	ND	2129.65	ND	0.163	5.29	0.657
23	CHK00-039	resident	Female	Adult	24-Feb-00	Centre Creek #1	0.007	ND	ND	ND	ND	13.771	ND	0.003	2135.34	0.153	0.256	4.86	0.919
24	CHK00-043	resident	Female	Adult	13-Mar-00	Lower Hatchery	0.010	0.113	ND	ND	29.73	8.700	ND	0.027	2668.74	ND	0.166	2.50	0.674
25	CHK00-044	resident	Female	Adult	13-Mar-00	Thurston	0.006	0.223	ND	ND	53.46	11.133	ND	ND	3391.38	ND	0.203	ND	1.542
26	CHK00-045	migrant	Female	Adult	15-Mar-00	Foley Creek	0.012	ND	11.743	ND	29.51	6.097	ND	0.018	2081.44	0.098	0.084	4.01	0.503
27	CHK00-048	resident	Female	Adult	24-Mar-00	Chwk River/Borden	0.011	ND	109.881	ND	405.07	1.196	ND	ND	3638.07	ND	0.193	97.25	3.022
28	CHK00-050	migrant	Female	Adult	19-Apr-00	Slesse Creek	0.009	ND	ND	ND	25.66	6.221	ND	0.027	3733.02	ND	ND	4.81	ND
29	CHK00-051	migrant	Female	Adult	9-May-00	Borden Creek	0.008	ND	145.951	ND	32.39	7.968	ND	0.005	3489.51	ND	0.129	34.82	1.381
30	CHK00-052	migrant	Female	Adult	10-May-00	Upper Tamihi	0.012	0.133	ND	ND	27.36	12.052	ND	0.334	3016.20	0.101	0.203	5.64	0.193
31	CHK00-053	resident	Female	Adult	16-May-00	Upper Hatchery	0.010	0.104	79.446	ND	43.70	4.340	ND	ND	2971.41	ND	ND	9.38	1.183
32	CHK00-056	resident	Female	Adult	24-May-00	Camp Foley	0.011	ND	ND	ND	108.43	1.488	ND	ND	2422.78	0.331	ND	1.89	1.077
33	CHK00-057	migrant	Female	Adult	30-May-00	Liumchen/Edwards	0.011	ND	ND	ND	66.76	5.840	ND	ND	2969.83	ND	ND	8.09	1.671
11	CHK99-202b	resident	Female	Adult	27-Jun-01	Hatchery	0.018	0.119	ND	ND	32.77	9.756	ND	ND	3932.71	0.092	0.131	11.38	ND
12	CHK99-208	migrant	Male	Adult	25-May-99	Lower Tamihi	0.010	0.138	ND	ND	23.99	6.250	ND	ND	3069.72	0.180	0.169	10.44	ND
13	CHK99-215	migrant	Female	Adult	2-Jun-99	Hatchery/Slesse	0.008	ND	ND	ND	ND	17.420	ND	ND	2325.29	0.138	0.195	22.34	0.350
14	CHK99-216b	resident	Female	Adult	29-May-01	Riverside	0.014	0.086	ND	12.286	ND	13.116	ND	ND	3263.85	0.194	0.124	17.87	ND
15	CHK99-222b	migrant	Female	Adult	7-Jun-01	Lower Tamihi #2	0.018	0.087	ND	ND	ND	10.813	ND	ND	3565.99	0.256	0.121	9.39	ND
34	CHK00-242	migrant	Male	Adult	17-Jun-00	Edwards	0.013	0.079	ND	ND	ND	14.426	ND	ND	2619.80	0.117	0.095	18.92	ND
35	CHK00-243	migrant	Male	Adult	18-Jan-00	Hatchery/Slesse	0.012	0.088	ND	ND	ND	9.559	ND	ND	1716.61	ND	ND	11.31	0.932
36	CHK00-256b	resident	Male	Adult	20-Feb-01	Chwk River/Tamihi	0.016	ND	ND	ND	41.64	3.752	ND	0.011	2941.11	0.385	ND	8.96	0.556
37	CHK00-259	migrant	Male	Adult	28-Jan-00	Anderson	0.015	ND	ND	ND	56.20	2.097	ND	0.003	2401.09	ND	ND	8.51	0.448
38	CHK00-270	resident	Male	Adult	3-Feb-00	Riverside	0.011	ND	ND	ND	21.16	8.541	ND	0.014	2558.55	ND	ND	13.43	0.581
39	CHK00-272	migrant	Male	Adult	4-Feb-00	Thurston	0.011	ND	ND	ND	ND	3.808	ND	0.011	3242.28	ND	0.103	13.41	0.379

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40	CHK00-275	resident	Male	Adult	7-Feb-00	Anderson 5	0.010	ND	ND	ND	ND	2.800	ND	0.005	2190.42	ND	0.103	11.11	1.043
41	CHK00-285	resident	Male	Adult	16-Feb-00	Allison Pool	0.011	ND	ND	ND	ND	2.223	ND	0.018	1367.67	ND	1.048	24.61	1.001
42	CHK00-293	resident	Male	Adult	8-Mar-00	Centre Creek #1	0.016	ND	ND	ND	35.71	1.336	ND	0.139	1854.75	ND	ND	10.21	ND
43	CHK00-294	migrant	Male	Adult	8-Mar-00	Riverside/Nesackwatc	0.012	ND	ND	ND	60.38	1.994	ND	ND	1494.61	ND	ND	8.64	0.648
44	CHK00-295	resident	Male	Adult	13-Mar-00	Lower Hatchery	0.010	ND	ND	ND	32.39	11.736	ND	ND	2608.48	ND	ND	12.36	0.821
45	CHK00-296	resident	Male	Adult	13-Mar-00	Thurston	0.010	ND	ND	ND	ND	1.489	ND	0.028	1537.12	ND	ND	7.66	1.152
46	CHK00-297	resident	Male	Adult	14-Mar-00	Anderson	0.018	ND	ND	ND	13.62	1.656	ND	ND	2353.15	ND	0.182	10.08	0.358
47	CHK00-298	migrant	Male	Adult	15-Mar-00	Foley Creek	0.015	ND	ND	ND	ND	1.386	ND	ND	2186.72	0.145	0.099	8.17	0.401
48	CHK00-299	resident	Male	Adult	16-Mar-00	Chwk River/Tamihi	0.009	ND	ND	ND	ND	3.144	ND	ND	3337.03	0.148	0.130	11.59	0.868
49	CHK00-300	migrant	Male	Adult	17-Mar-00	Upper Tamihi	0.010	ND	ND	ND	ND	2.464	ND	0.012	2213.16	0.109	ND	11.40	1.300
50	CHK00-301	resident	Female	Adult	23-Mar-00	Anderson	0.012	ND	ND	ND	ND	2.577	ND	ND	2575.56	0.093	0.125	14.54	0.570
51	CHK00-302	resident	Male	Adult	24-Mar-00	Little Tamihi 2	0.012	ND	ND	ND	ND	12.044	ND	ND	2193.81	ND	ND	7.62	0.863
52	CHK00-304	resident	Male	Adult	30-Mar-00	Chwk River/Borden	0.014	ND	ND	ND	ND	2.029	ND	0.001	1848.43	ND	ND	7.00	0.796
53	CHK00-316b	resident	Male	Adult	22-May-01	Hatchery/Slesse #1	0.020	ND	ND	ND	ND	1.784	ND	ND	2748.33	ND	ND	11.15	0.282
54	CHK00-318b	migrant	Female	Adult	27-Feb-01	Tamihi	0.017	ND	ND	ND	ND	2.535	ND	0.135	2585.52	ND	0.071	9.75	0.135
55	CHK00-336b	migrant	Male	Adult	20-Jun-01	Tamihi	0.017	ND	ND	ND	ND	3.351	ND	ND	2903.57	0.084	ND	11.60	0.021
56	CHK00-339b	resident	Male	Adult	12-Jun-01	Chwk River/Tamihi	0.023	ND	ND	ND	68.14	1.673	ND	ND	2636.33	0.063	0.009	15.11	ND
57	CHK00-364b	resident	Female	Adult	1-Mar-01	Vedder	0.015	ND	ND	ND	31.23	2.033	ND	0.077	2459.73	ND	ND	9.85	ND
58	CHK00-399	migrant	Male	Adult	9-May-00	Borden Creek	0.011	ND	247.537	ND	144.28	0.813	ND	ND	1983.73	0.120	0.045	8.45	ND
59	CHK00-410	resident	Female	Adult	16-May-00	Eagle Roost	0.011	ND	ND	ND	157.25	2.525	ND	0.109	3058.97	ND	ND	12.88	ND
60	CHK00-445	resident	Male	Adult	19-Jun-00	Eagle Roost	0.012	ND	ND	ND	125.53	2.157	ND	ND	2638.26	ND	ND	15.80	ND
61	CHK01-065	resident	Female	Adult	27-Feb-01	Chwk River/Tamihi	0.012	ND	ND	ND	76.84	4.679	ND	ND	3213.12	0.158	ND	73.08	ND
62	CHK01-066	resident	Female	Adult	27-Feb-01	Allison Pool	0.012	ND	ND	ND	103.61	4.779	ND	0.089	2380.51	0.273	ND	10.15	0.883
63	CHK01-067	resident	Female	Adult	21-Mar-01	Anderson	0.020	ND	ND	ND	87.24	15.289	ND	ND	6517.93	0.539	ND	8.80	0.238
64	CHK01-068	resident	Female	Adult	22-Mar-01	Chwk River/Tamihi	0.016	ND	ND	ND	88.75	1.983	ND	0.071	2911.74	0.156	ND	12.12	0.685
65	CHK01-070	resident	Female	Adult	10-Apr-01	Thurston Correctional	0.013	ND	ND	ND	113.88	22.202	ND	0.095	2402.33	0.164	ND	10.82	0.278
66	CHK01-071	resident	Female	Adult	10-Apr-01	Centre Creek #1	0.015	ND	ND	ND	55.19	2.462	ND	0.284	3575.10	0.267	0.019	13.13	ND
67	CHK01-072	migrant	Female	Adult	3-May-01	Liumchen	0.013	ND	ND	ND	82.77	7.568	ND	ND	6698.39	0.284	0.063	8.26	0.996
68	CHK01-073	migrant	Female	Adult	7-May-01	Upper Tamihi	0.015	ND	ND	ND	60.95	2.383	ND	0.072	2419.27	ND	ND	8.72	0.347
69	CHK01-076	migrant	Female	Adult	1-Jun-01	Foley Creek	0.016	ND	ND	ND	103.88	3.163	ND	ND	3248.28	0.211	0.028	6.16	0.658
70	CHK01-077	migrant	Female	Adult	20-Jun-01	Liumchen Creek	0.012	0.115	ND	ND	115.23	14.683	ND	0.107	8344.99	0.126	0.250	10.67	1.057
71	CHK01-458	migrant	Male	Adult	21-Feb-01	Thurston	0.015	ND	ND	ND	122.23	1.533	ND	0.078	3083.50	ND	ND	9.11	ND
72	CHK01-463	migrant	Male	Adult	27-Feb-01	Tamihi	0.013	ND	ND	ND	143.00	5.101	ND	0.088	3186.09	ND	ND	16.37	ND
73	CHK01-464	resident	Male	Adult	27-Feb-01	Allison Pool	0.018	ND	ND	ND	118.85	1.286	ND	0.071	2571.50	ND	ND	6.38	ND
74	CHK01-470	resident	Male	Adult	21-Mar-01	Anderson	0.020	ND	23.917	ND	95.58	6.304	ND	0.058	3467.00	0.221	0.072	11.31	ND
75	CHK01-472	migrant	Male	Adult	29-Mar-01	Lower Tamihi	0.015	ND	ND	ND	115.06	1.667	ND	0.069	2706.49	ND	ND	7.62	ND
76	CHK01-473	migrant	Male	Adult	29-Mar-01	Liumchen	0.018	ND	ND	ND	135.81	2.050	ND	ND	2390.69	ND	ND	10.57	0.095
77	CHK01-474	resident	Male	Adult	30-Mar-01	Upper Hatchery	0.014	ND	ND	ND	181.28	0.617	ND	ND	2973.77	ND	ND	9.35	ND
78	CHK01-475	resident	Male	Adult	3-Apr-01	Upper Hatchery	0.016	ND	ND	ND	148.74	7.058	ND	0.064	3041.93	ND	ND	11.06	ND
79	CHK01-504	resident	Male	Adult	15-May-01	Edwards	0.020	ND	ND	ND	86.72	2.809	ND	ND	2999.80	ND	ND	10.43	ND
80	CHK01-514	migrant	Male	Adult	28-May-01	Slesse Creek	0.020	ND	ND	ND	78.49	6.708	ND	ND	2326.36	0.074	ND	13.71	ND
81	CHK01-535	migrant	Female	Adult	3-Jul-01	Foley Creek	0.018	ND	ND	ND	74.18	1.839	ND	ND	2937.18	0.076	0.010	10.23	ND
82	CHK01-536	migrant	Male	Adult	5-Jul-01	Upper Tamihi	0.021	ND	ND	ND	68.71	4.413	ND	ND	2896.91	0.059	ND	21.45	ND

*Detection Limit (DL) in mg/kg dw based on avg. sample weight of 0.013 g.

Appendix 4

Feather ICP-MS analysis: Concentrations in mg/kg dry weight (ppm)

Chilliwack watershed American Dipper feather samples

Sample	Fe	Hg	K	Li	Mg	Mn	Mo	Na	Ni	P	Pb	Sb	Se	Sn	Sr	Te	Tl	Tl	U	V	Zn	Zr
***DL(ppm)**	**7.69**	**0.04**	**38.46**	**0.77**	**7.69**	**0.08**	**0.08**	**7.69**	**1.54**	**15.38**	**0.08**	**0.08**	**2.31**	**0.08**	**0.08**	**0.15**	**0.77**	**0.08**	**0.08**	**7.69**	**0.77**	**0.08**
1	59.22	0.712	99.07	0.611	1159.23	1.028	ND	1323.52	2.734	106.01	0.348	3.775	6.235	0.148	2.489	ND	0.466	ND	ND	ND	132.13	ND
2	ND	0.376	318.04	0.836	1216.85	1.777	ND	1213.51	1.545	67.53	6.782	6.337	4.170	0.098	6.383	ND	0.517	ND	ND	ND	130.08	ND
3	169.90	0.847	243.28	0.637	1378.35	2.190	0.0633	2040.53	2.681	114.03	1.721	8.463	3.906	0.190	3.226	ND	1.275	ND	ND	5.583	139.11	ND
4	ND	0.370	118.10	0.634	1803.02	0.602	ND	1905.40	ND	82.20	0.006	7.452	4.077	0.035	2.719	ND	ND	ND	ND	ND	144.77	ND
5	ND	1.257	196.38	ND	807.63	1.200	ND	1119.21	ND	167.99	ND	0.802	7.390	0.062	8.440	ND	ND	ND	ND	ND	178.94	ND
6	79.07	1.184	158.96	1.515	730.62	0.599	ND	954.76	ND	145.57	ND	0.453	6.041	0.070	3.779	ND	0.818	ND	ND	12.749	165.87	ND
7	26.33	1.329	1923.54	0.781	2076.27	0.426	ND	3123.84	1.494	167.41	0.549	0.991	4.719	0.151	1.860	ND	2.839	ND	ND	ND	250.36	0.078
8	534.18	0.612	176.95	ND	589.48	0.496	ND	960.66	ND	401.25	0.307	0.438	5.702	0.087	4.698	ND	152.867	ND	ND	ND	121.57	0.061
9	ND	0.382	75.72	1.042	1086.81	0.189	ND	1432.22	ND	94.82	ND	0.030	ND	0.079	1.929	ND	ND	ND	ND	ND	128.96	ND
10	ND	0.630	ND	1.048	1061.71	ND	ND	1606.39	ND	45.34	ND	0.210	3.417	0.216	1.459	ND	ND	ND	ND	ND	143.14	ND
16	ND	0.183	69.76	ND	921.45	0.393	ND	1012.44	ND	87.63	0.625	1.625	ND	0.002	0.960	ND	ND	ND	ND	ND	138.90	ND
17	ND	1.463	ND	1.039	1558.79	0.433	ND	1387.47	ND	117.08	1.693	0.457	ND	0.048	1.317	ND	21.735	ND	ND	ND	129.41	ND
18	ND	0.801	78.35	ND	1368.89	0.892	ND	1342.77	18.497	172.75	0.731	0.425	6.949	ND	2.588	ND	ND	ND	ND	ND	142.32	0.299
19	ND	0.304	179.10	ND	800.77	0.707	ND	861.71	ND	52.77	0.088	0.681	8.588	ND	5.259	ND	2.691	ND	ND	ND	156.10	ND
20	ND	0.123	118.86	ND	1230.16	0.787	ND	1466.30	ND	148.95	0.956	1.756	ND	0.058	1.816	ND	ND	ND	ND	ND	129.24	ND
21	ND	1.145	52.64	ND	934.17	0.524	ND	1221.36	2.250	90.50	0.046	2.068	3.182	0.046	1.970	ND	ND	ND	ND	ND	114.08	0.141
22	2479.18	0.494	53.86	ND	731.04	0.946	ND	1000.39	ND	845.10	ND	0.547	6.934	0.051	1.608	ND	108.389	ND	ND	ND	103.67	0.780
23	2.31	ND	ND	ND	900.01	1.286	ND	944.06	ND	109.23	0.404	0.359	5.629	ND	1.209	ND	ND	ND	ND	ND	119.03	ND
24	374.45	0.632	93.51	ND	1221.36	0.031	ND	1400.43	ND	268.13	0.009	1.253	5.450	ND	1.260	ND	39.929	ND	ND	ND	134.14	ND
25	ND	0.058	132.73	ND	1231.93	0.330	ND	1511.32	ND	159.76	ND	0.455	9.350	ND	4.187	ND	ND	ND	ND	ND	178.02	ND
26	ND	1.046	ND	ND	1131.73	0.112	ND	479.59	ND	72.81	0.902	0.189	5.688	ND	1.029	ND	ND	ND	ND	ND	111.07	ND
27	124.86	0.442	218.09	1.151	1386.00	5.535	ND	3233.83	4.295	110.28	2.272	0.130	7.255	0.001	3.111	ND	1.540	ND	ND	9.782	136.72	1.348
28	34.22	0.307	265.30	ND	904.59	1.429	ND	357.66	3.206	134.39	0.257	0.489	7.594	ND	8.006	ND	38.773	ND	ND	11.524	135.44	ND
29	ND	0.419	453.28	ND	788.72	1.561	ND	1259.15	ND	81.44	0.835	0.626	ND	ND	4.128	ND	ND	ND	ND	ND	133.50	ND
30	95.46	0.495	203.65	ND	937.08	0.727	ND	941.67	ND	155.17	ND	0.255	4.863	ND	3.661	ND	0.037	0.168	ND	ND	122.42	ND
31	ND	0.667	75.47	ND	1324.37	1.004	ND	799.23	ND	88.14	ND	0.556	ND	ND	2.897	ND	ND	ND	ND	10.315	146.15	ND
32	ND	0.821	64.86	ND	760.37	1.044	ND	1444.67	ND	103.96	5.949	0.368	3.509	ND	1.171	ND	ND	ND	ND	9.662	129.60	ND
33	ND	0.576	ND	ND	1001.99	0.467	ND	1228.72	ND	36.18	0.424	0.035	4.826	ND	5.666	ND	ND	ND	ND	9.448	149.81	ND
11	260.48	0.798	209.22	ND	1632.78	6.158	ND	2354.63	ND	119.46	1.032	16.322	10.597	0.154	4.918	ND	1.302	ND	ND	5.896	149.96	0.083
12	233.88	0.440	59.86	1.054	1098.85	0.454	ND	1935.91	ND	114.59	1.248	28.917	4.348	0.260	4.036	ND	ND	ND	ND	10.019	132.31	0.164
13	196.86	0.618	111.86	1.485	1010.22	0.251	ND	1666.97	ND	96.80	0.693	35.138	4.389	0.374	2.153	ND	ND	ND	ND	ND	143.90	0.410
14	244.77	0.910	124.91	ND	1544.84	3.364	ND	1996.07	ND	87.22	0.794	15.986	4.066	0.179	2.359	ND	3.206	ND	ND	ND	152.22	0.567
15	176.18	0.633	188.65	ND	1480.18	2.367	ND	1824.90	1.234	118.38	0.667	7.128	5.732	0.173	4.318	ND	1.592	ND	ND	6.081	153.64	0.173
34	293.78	0.896	ND	ND	1620.14	0.805	ND	1861.10	ND	122.99	4.250	30.025	5.678	0.180	3.096	ND	ND	ND	ND	8.015	135.71	0.071
35	ND	0.545	ND	ND	912.18	0.118	ND	998.93	ND	59.90	0.458	4.918	5.905	0.119	2.963	ND	ND	ND	ND	ND	148.10	ND
36	ND	0.388	0.34	ND	977.48	0.826	ND	1319.62	ND	43.41	0.337	3.404	3.842	0.124	3.399	ND	ND	ND	ND	7.244	111.76	0.306
37	104.15	0.538	ND	ND	1165.31	0.304	ND	1091.81	ND	98.35	5.200	63.688	4.048	0.103	1.768	ND	ND	ND	ND	6.993	108.65	0.259
38	ND	0.262	40.14	ND	1145.09	ND	ND	1521.32	ND	81.43	0.574	8.603	6.989	0.185	2.631	ND	ND	ND	ND	10.459	159.60	0.136
39	177.79	0.165	82.55	ND	1146.30	0.319	ND	1249.33	ND	124.76	1.533	3.066	7.075	0.218	3.863	ND	ND	ND	ND	10.057	174.43	0.495

*Detection Limit (DL) in mg/kg dw based on avg. sample weight of 0.013 g.

Appendix 4

40	87.60	0.433	ND	ND	785.43	ND	ND	1622.67	27.397	159.16	0.827	3.807	3.718	0.213	2.120	ND	ND	ND	ND	10.426	111.72	ND
41	95.10	ND	ND	ND	664.75	0.425	ND	1065.62	872.995	56.17	0.474	2.706	6.557	1.115	2.575	ND	ND	ND	ND	9.689	106.76	ND
42	6.66	0.433	ND	ND	752.08	0.031	ND	924.30	ND	73.57	0.265	1.780	2.176	0.108	1.341	ND	ND	0.076	ND	ND	116.60	0.071
43	53.10	0.330	1.26	ND	825.30	ND	ND	1130.12	ND	62.84	0.189	2.413	ND	0.150	1.123	ND	ND	ND	ND	8.629	108.62	0.394
44	68.89	0.490	ND	ND	1193.28	0.035	ND	1345.53	ND	76.61	2.030	2.288	3.558	0.151	1.760	ND	ND	ND	ND	10.190	126.91	ND
45	32.19	0.127	15.17	ND	692.29	ND	ND	1099.53	ND	62.60	0.498	1.614	5.307	0.149	1.321	ND	ND	ND	ND	10.264	106.30	0.029
46	140.29	0.437	17.63	ND	1126.99	2.314	ND	1202.67	ND	81.77	0.334	2.226	2.813	0.077	2.285	ND	ND	ND	ND	5.845	112.03	0.107
47	90.52	0.195	21.99	ND	1044.11	ND	ND	1431.57	ND	120.66	0.751	74.002	5.859	0.104	1.579	ND	ND	ND	ND	7.507	109.06	0.152
48	210.30	1.326	45.61	ND	1328.32	0.680	ND	1543.36	ND	235.44	2.312	133.967	7.716	0.179	2.383	ND	ND	ND	ND	12.636	152.18	0.009
49	10.86	0.170	16.78	ND	802.27	ND	ND	961.09	ND	77.44	0.768	0.890	4.282	0.102	4.068	ND	ND	ND	ND	10.154	113.61	0.530
50	133.82	0.203	ND	ND	1323.97	0.291	ND	1286.87	ND	90.11	0.745	0.505	8.958	0.112	3.647	ND	ND	ND	ND	9.221	103.16	0.012
51	99.84	0.325	1.75	ND	1225.69	0.068	ND	1308.05	ND	70.56	0.438	1.157	4.960	0.158	1.763	ND	ND	ND	ND	9.153	107.76	ND
52	36.01	0.738	8.98	ND	905.57	ND	ND	1314.97	ND	65.92	0.213	16.747	4.666	ND	1.171	ND	ND	ND	ND	7.966	104.60	0.189
53	131.16	1.029	35.09	ND	1413.28	0.767	ND	1293.86	ND	75.20	1.076	0.776	4.954	0.082	2.218	ND	ND	ND	ND	5.792	113.16	0.147
54	65.11	0.278	15.76	ND	1049.20	1.694	ND	1097.02	ND	76.19	0.898	10.725	7.211	1.232	2.916	ND	1.537	0.091	ND	ND	102.28	0.007
55	ND	0.284	22.08	ND	1283.00	0.654	ND	866.12	ND	95.37	0.969	0.194	7.765	0.126	4.200	ND	ND	ND	ND	ND	132.24	ND
56	161.63	1.440	ND	ND	1485.49	1.223	ND	1334.05	1.642	101.15	1.177	26.392	4.988	0.143	1.561	ND	1.914	ND	ND	5.088	124.95	ND
57	64.14	0.665	ND	ND	1026.44	0.775	ND	774.91	ND	93.13	1.443	1.279	6.676	0.109	2.794	ND	0.739	ND	ND	7.646	147.59	ND
58	ND	ND	ND	ND	1007.28	1.170	ND	1301.60	ND	84.96	0.903	10.221	5.054	0.146	2.125	ND	ND	ND	ND	9.231	123.63	ND
59	126.50	0.196	15.60	ND	1321.54	1.726	ND	1553.36	ND	132.10	0.330	1.365	7.170	0.105	2.639	ND	ND	ND	ND	10.123	144.22	ND
60	141.89	0.823	ND	ND	991.99	0.778	ND	1454.92	ND	117.40	0.425	1.448	7.274	0.132	2.543	ND	ND	ND	ND	9.025	128.51	ND
61	195.13	0.571	482.54	ND	1307.48	2.015	ND	1529.10	ND	144.11	0.853	6.989	7.973	0.757	6.058	ND	ND	ND	ND	10.086	133.24	ND
62	147.32	0.543	ND	0.852	1419.60	0.478	ND	1149.16	ND	108.73	0.380	0.161	9.366	0.119	3.250	ND	ND	ND	ND	9.795	122.31	ND
63	65.21	0.729	ND	ND	1491.16	1.852	ND	1144.57	ND	100.17	0.761	0.137	6.978	0.086	15.750	ND	1.002	ND	ND	5.709	128.50	ND
64	93.91	0.178	ND	ND	1628.29	1.791	ND	1505.95	ND	85.08	0.891	0.086	7.973	0.102	4.486	ND	ND	ND	ND	7.223	127.07	ND
65	ND	0.527	ND	ND	1466.11	0.611	ND	1585.99	ND	82.63	0.552	0.567	3.415	0.185	1.916	ND	ND	ND	ND	9.103	141.77	ND
66	77.70	0.392	ND	ND	1628.33	0.505	ND	1435.59	ND	115.23	0.615	0.371	7.861	0.151	1.885	ND	ND	0.113	ND	ND	128.43	ND
67	72.54	0.525	132.28	ND	1573.78	1.729	ND	1292.76	ND	100.56	0.392	0.184	5.710	0.260	20.262	ND	ND	ND	ND	ND	150.35	ND
68	35.17	0.335	ND	ND	1014.27	0.416	ND	884.68	ND	97.18	ND	0.300	5.177	0.079	3.225	ND	ND	ND	ND	7.151	107.12	ND
69	69.95	0.301	220.35	ND	1214.63	2.246	ND	1204.55	ND	118.23	0.148	0.856	9.314	0.118	6.228	ND	ND	ND	ND	6.941	128.20	ND
70	117.66	1.093	153.88	ND	1715.07	9.820	ND	1444.54	ND	138.15	0.658	0.341	12.488	0.141	32.993	ND	2.498	ND	ND	9.616	164.18	ND
71	99.98	0.190	ND	ND	1778.79	0.196	ND	1168.15	ND	125.58	ND	10.826	9.741	0.174	3.177	ND	ND	ND	ND	7.333	124.12	ND
72	442.78	0.422	ND	ND	1732.64	1.391	ND	1248.62	ND	101.85	0.476	1.196	5.839	0.146	2.913	ND	ND	ND	ND	8.629	123.23	ND
73	61.94	0.566	ND	ND	1690.08	0.047	ND	1367.00	ND	117.80	0.136	0.848	6.784	0.119	1.372	ND	ND	ND	ND	6.407	119.12	ND
74	209.01	1.249	27.94	ND	1314.51	7.037	ND	987.95	2.170	135.68	1.087	1.051	6.647	0.097	6.373	ND	4.251	ND	ND	5.858	137.19	ND
75	118.64	1.281	ND	ND	1351.60	0.180	ND	1088.86	ND	113.96	0.436	0.995	7.737	0.117	1.767	ND	ND	ND	ND	7.843	109.92	ND
76	46.37	0.383	ND	ND	1028.72	0.248	ND	989.66	ND	82.35	0.294	0.812	4.078	0.109	2.657	ND	ND	ND	ND	5.797	112.17	ND
77	135.50	1.112	ND	ND	1970.41	0.356	ND	1578.46	ND	129.72	0.499	1.210	13.329	0.130	2.261	ND	2.410	0.123	ND	ND	145.34	ND
78	135.44	1.338	11.36	ND	1455.53	0.283	ND	1138.19	ND	123.66	0.462	2.089	4.654	0.071	2.641	ND	2.379	ND	ND	ND	130.41	ND
79	42.66	0.490	23.87	ND	1985.03	1.256	0.0685	1048.71	ND	89.91	0.405	2.346	4.605	0.104	1.673	ND	1.828	ND	ND	5.521	121.09	ND
80	97.72	0.400	ND	ND	1198.63	0.478	ND	994.42	ND	95.34	0.696	8.631	4.018	0.096	1.839	ND	2.946	ND	ND	ND	121.78	ND
81	128.82	0.287	55.91	ND	1433.28	1.764	ND	1154.42	2.618	118.97	0.884	0.906	4.637	0.147	2.676	ND	3.400	ND	ND	ND	121.36	ND
82	47.01	0.383	ND	ND	1253.04	0.633	ND	882.56	ND	84.82	0.485	0.885	4.059	0.131	6.761	ND	1.740	ND	ND	ND	97.38	ND

Appendix 4

Feather Hg analysis 1999: Concentrations in mg/kg dry weight (ppm)								
Chilliwack watershed American Dipper feather samples								
Sample	**Bird ID/Sample ID**	**Group**	**Sex**	**Age**	**Date Collected**	**Location**	**Sample dry Weight (g)**	**Hg**
Detect limit (ppm)								**0.04**
83	CHK99-004	resident	Female	Adult	18-Feb-99	Anderson	NA	0.727
84	CHK99-011	resident	Female	Adult	11-Mar-99	Tamihi Creek	NA	0.928
85	CHK99-012	resident	Female	Adult	29-Mar-99	Chwk River/Tamihi	NA	0.428
86	CHK99-013	migrant	Female	Adult	29-Jul-99	Upper Tamihi Creek	NA	0.419
87	CHK99-105	migrant	Male	Adult	16-Feb-99	Hatchery	NA	0.754
88	CHK99-107	migrant	Female	Adult	17-Feb-99	Anderson	NA	0.583
89	CHK99-115	migrant	Male	Adult	24-Feb-99	Edwards	NA	0.653
90	CHK99-117	resident	Male	Adult	25-Feb-99	Vedder	NA	0.791
91	CHK99-119	unknown	Male	Adult	26-Feb-99	Vedder	NA	1.005
92	CHK99-124	resident	Male	Adult	02-Mar-99	Thurston	NA	0.804
93	CHK99-127	unknown	Female	Adult	03-Mar-99	Thurston	NA	1.468
94	CHK99-130	migrant	Male	Adult	04-Mar-99	Camp Foley	NA	0.957
95	CHK99-149	resident	Male	Adult	06-Apr-99	Chwk Lake	NA	0.884
96	CHK99-167		Unknown	Juvenile	05-May-99	Anderson #4	NA	1.206
97	CHK99-169	resident	Male	Adult	05-May-99	Anderson #4	NA	1.176
98	CHK99-201	resident	Male	Adult	19-May-99	Hatchery/Slesse Creek	NA	2.088
99	CHK99-202	resident	Female	Adult	19-May-99	Hatchery	NA	2.741
100	CHK99-203	resident	Male	Adult	20-May-99	Skesse Creek #2	NA	1.348
101	CHK99-214	resident	Male	Adult	01-Jun-99	Anderson #5	NA	1.570
102	CHK99-216	resident	Female	Adult	04-Jun-99	Riverside	NA	1.766
103	CHK99-218	migrant	Male	Adult	08-Jun-99	Foley Creek #2	NA	1.211
104	CHK99-219		Unknown	Juvenile	09-Jun-99	Allison Pool	NA	1.099
105	CHK99-226		Unknown	Juvenile	11-Jun-99	Chwk River/Borden Cre	NA	0.815
106	CHK99-227		Unknown	Juvenile	17-Jun-99	Foley Creek	NA	0.359
107	CHK99-229	migrant	Female	Adult	17-Jun-99	Foley Creek #4	NA	0.701
108	CHK99-232	migrant	Female	Adult	25-Jun-99	Slesse Creek #4	NA	0.702

*Detection Limit (DL) in mg/kg dw based on avg. sample weight of 0.013 g.

CHAPTER 7:

SYNTHESIS AND CONCLUSIONS

Migratory patterns and its implications

It was clear that the majority of our dipper population was highly mobile, producing large fluctuations in densities during the annual cycle. In addition, two distinct groups existed within the population that had different migratory strategies. Resident dippers occupied territories year-round along the main river channel with little or no movement between seasons. Migrants, however, made up the majority of the population and these individuals typically concentrated on the low elevation river during winter then made short altitudinal movements upstream onto the tributaries in the breeding season. This is the first study to confirm the presence and follow the movements of resident and altitudinal migrant American dippers within a single population. Since these two groups occupy adjacent habitats that potentially differ in available resources (food and nest sites) and in sources of contaminants, it is essential to understand how populations use these habitats during an annual cycle.

Many ecologists recognize that individuals and populations of animals occupying neighboring habitats or patches are frequently related. For example, seminal work by Krebs (1971) on great tits in southern England, demonstrated that tits occupying the mixed deciduous woodland had higher reproductive success in this habitat but excluded other birds into "sub-optimal" adjacent hedgerows. Hedgerows, however, provided a buffer against fluctuations in woodland density by replacing individuals in the optimal habitat whenever opportunities arose. Just as the woodlands seemed to be better breeding habitat for great tits, resident dippers on the river similarly appeared to benefit from earlier breeding, greater numbers of second broods and higher productivity than migrants on creeks. However, differences in elevation, habitat and nesting substrates between the

river and creeks had no direct influence on overall breeding performance. Instead, the effect was primarily due to timing of breeding, which in creek birds was delayed by the migration period. Surplus birds wintering on the low elevation river are forced to move in spring onto the tributaries to gain access to limited breeding sites. Therefore, as in the case of great tits in hedges and woodlands, knowledge of how and why populations distribute themselves amongst patches or locations is critical to the interpretation of habitat quality.

The observed difference in migratory strategies among resident and migrant dippers in the study watershed not only had implications for habitat use and reproductive performance, but was also found to influence diet and contaminant exposure. Stable isotope analysis revealed salmon fry prey made up a significantly larger portion of the river residents' diet compared to creek migrants, likely as a result of differences in prey availability between the river and creeks. Other studies have similarly shown that even small-scale differences in breeding locations can result in differences in diet and consequently contaminant burdens (Hebert et al. 1997, Bustnes et al. 2000, Bearhop et al. 2000). Therefore, future ecotoxicology studies should evaluate the potential effects of small-scale periodic movements on contaminant exposure.

Who stays, who goes?

Throughout the study, I have frequently asked the question: who stays and who goes? In other words, which birds remain resident and which birds become migrants? There is considerable evidence that breeding densities in birds are limited partly by nest site availability or by spacing behavior (Lack 1954, Klomp 1972, Newton 1976). Evidence of surplus breeding adults has been shown through removal experiments and

addition of nest-boxes (van Balen et al. 1982, Village 1983, Brawn and Balda 1988, Pedersen 1988). The presence of large numbers of dippers on low elevation rivers in winter and their absence in spring suggests the non-breeding population exceeds the breeding carrying capacity of the river, thus initiating the spring migration (Campbell et al. 1997). Many studies assume that the surplus breeders are young birds that are inexperienced in establishing territories. However, only through intensive mark-recapture studies is the structure of the population identified.

Hundreds of observations of banded dippers in my study area in addition to formal population models did not identify a specific subset of the population as the source of divergence in migratory strategies. First year birds recruited into the breeding population equally as residents or migrants, regardless of sex. Since juveniles born on the river or a creek established breeding territories in their first year regardless of natal origins, genetics or natal site also did not predict where individuals would recruit into the population as adults. Therefore, the process that determines migratory strategy in dippers is still unknown. However, the technique of marking individuals proved essential in excluding the hypotheses of age, sex, and natal site biases existing among resident and migrant breeders.

Residents versus migrants: who's doing better?

Preliminary investigation of data on the reproductive success of resident and migrant dippers suggested that residents were at an advantage to migrants in terms of productivity. Furthermore, river residents had a greater availability of fish prey to supplement their diet. However, I found no evidence that residents had greater ecological fitness since residents and migrants reproduced equally well when the influence of timing

of breeding was removed. Insufficient data on adult or juvenile survival of residents and migrants prevented the determination of whether any long-term fitness effects exist from differential migration strategies.

The toxicological results presented conflicting profiles of residents and migrants depending on whether organic or metal exposure was assessed. For organochlorines, PCBs, and mercury, resident dippers had greater exposure to these contaminants largely through their higher fish diet. While metal exposure was more significant for migrant birds consuming mainly insect diets. In spite of these differences, reproductive performance was similar between residents and migrants and the contaminant levels observed in this study were generally below toxicity thresholds. Therefore, we have no evidence that contaminants were contributing to overt effects on resident and migrant dippers in this watershed.

Implications for using the American dipper as a biomonitor

Avian ecologists now recognize that contaminants are an important factor in population declines (Newton 1998, Fox 2000). The effects of contaminants may be indirect, delayed or widespread in nature, which can confound the difficulty in interpreting their impact considering other environmental factors that influence populations. Therefore, there is a general need to improve the design of field studies to incorporate toxicological, demographic and ecological data to accurately assess risks to populations. Furthermore, non-destructive sampling of blood, feathers or eggs can be used as effective biomarkers, which can permit us to gain additional information on individuals through marking and telemetry methods. Finally, long-term studies are equally essential in understanding the impacts of pollutants on individuals and

ecosystems since they incorporate natural fluctuations that can mask contaminant effects in the short term. This research can therefore be used as a model for incorporating toxicological and demographic data over multiple years, to attempt to improve the quality of interpretation of toxicological effects on a species.

Altitudinal migration to the breeding site caused a considerable redistribution of the population during the annual cycle. This has implications for using dippers as biomonitors of habitat quality or stream condition. Given the large fluctuations in numbers and predictable patterns of movement, censuses must be carefully timed to assess population size and relative changes in density among sites and years. In addition, migratory status can further influence timing of breeding and observed productivity.

Local conditions were not found to be homogeneous within the watershed with residents and migrants experiencing different contaminant levels as a result of breeding location and diet effects. However, dipper eggs reflected recent dietary uptake and local contaminant sources proving they were useful tools for biomonitoring trace organic contaminants. Nestlings should similarly be representative of the territory but may not accumulate significant concentrations during the nestling period to warrant their use in some toxicology studies. Non-destructive sampling of feathers is also a valuable tool to detect trends in metal profiles; however, we could not accurately determine whether molting location differed from the breeding site. Therefore, caution in interpreting feather concentrations of altitudinal migrants and residents are necessary.

Collections of prey items including benthic invertebrates and salmon fry can be useful in determining exposure risk to aquatic consumers, particularly for trace metals. I recommend using models that incorporate the effects of body mass and average daily

intake to improve our assessments of risk to wildlife consumers. Prey samples often represent local conditions, and have the advantage of being easier to collect than bird samples. However, lower trophic level invertebrates often do not accumulate organic contaminants to sufficient levels to permit analytical detection, nor do they represent an average exposure to higher organisms that are longer-lived and more mobile. Therefore, use of prey samples may be beneficial for initial assessments, but should be followed by intensive monitoring programs using wildlife consumers.

Relevance to conservation

Conservation efforts often can protect only small areas due to limited resources, despite the fact that animals can move routinely between habitats. While many areas of watersheds may appear unsuitable for dippers, this research highlights the fact that dipper habitat requirements can change during the annual cycle. For example, good wintering areas on low elevation, low flow streams may not be suitable to sustain significant breeding populations due to lack of suitable nest sites, while breeding sites on high elevation tributaries may not be suitable for wintering due to ice cover or extreme temperature fluctuations. Addition of platforms under bridges or nest boxes in critical wintering areas may augment nest site availability to increase the number of breeders on low elevation rivers. However, watersheds are inherently continuous, such that alterations in water quality in headwater streams ultimately affect downstream sites. Contaminants originating from long-range atmospheric sources were found to bioaccumulate and biomagnify in lotic food chains primarily affecting the downstream, larger drainage area sites. Therefore, evidence from this study suggests it is essential to

consider dipper habitat protection on a watershed scale to effectively conserve stable populations.

Future research

Future research to determine what variables, including nest site availability or intraspecific competition, are important in influencing individual migration strategies in dippers would improve our current understanding of the natural history of this species. These hypotheses could be tested with the addition of nest boxes in high-density wintering areas to determine whether the breeding population increases in response to the augmented nest site availability and reduced competition. Other research that addresses key questions about survival and reproductive costs for permanent residents or altitudinal migrants may also shed light on our current understanding of altitudinal migration.

Using the methods described in chapter 2, I recommend additional censuses of American dippers to determine population size and long-term trends in population density for impacted and non-impacted watersheds throughout the province. Since we have few estimates from current survey methods in this province and across the species range, this will allow for more accurate assessments of the population's stability over its entire geographic range. Further investigations into the relationships between dipper populations and other riverine species would also allow us to identify key differences in habitat use that may be important for monitoring habitat or stream condition.

Literature Cited

Bearhop, S., Waldron, S., Thompson, D. and Furness, R. 2000. Bioamplification of mercury in the Great Skua *Catharacta skua* chicks: the influence of trophic status as determined by stable isotope signatures of blood and feathers. Marine Pollution Bulletin 40: 181-185.

Brawn, J.D. and Balda, R.P. 1988. Population biology of cavity nesters in northern Arizona: do nest sites limit breeding densities? Condor 90: 61-71.

Bustnes, J.O., Erikstad, K.E., Bakken, V., Mehlum, F. and Skaare, J.U. 2000. Feeding ecology and the concentration of organochlorines in Glaucous gulls. Ecotoxicology 9: 179-186.

Campbell, R. W., N. K. Dawe, I. McTaggart-Cowan, J. M. Cooper, G.W. Kaiser, M. C. E. McNall and G. E. J. Smith. 1997. American Dipper (*Cinclus mexicanus swainson*). Pp. 336-343 *In:* The Birds of British Columbia, Vol. 3. UBC Press, Vancouver, BC.

Fox, G.A. 2000. Perturbations in terrestrial vertebrate populations: contaminants as a cause. Pp. 19-59. *In:* Environmental contaminants and terrestrial vertebrates: effects on populations, communities, and ecosystems. (*Eds.* P. Albers, G. Heinz, and H. Olendorf). SETAC special publication, Pensacola, FL.

Hebert, C.E., Shutt, J.L. and Norstrom, R.J. 1997. Dietary changes cause temporal fluctuations in polychlorinated biphenyl levels in Herring gull eggs from Lake Ontario. Environmental Science and Technology 31: 1012-1017.

Klomp, H. 1972. Regulation of the size of bird populations by territorial behaviour. Netherlands Journal of Zoology 22: 456-488.

Krebs, J.R. 1971. Territory and breeding density in the Great Tit, *Parus major*. Ecology 52: 2-22.

Lack, D. 1954. The natural regulation of animal numbers. Clarendon Press, Oxford.

Newton, I. 1976. Population limitation in diurnal raptors. Canadian Field Naturalist 90: 274-300.

Newton, I. 1998. Population limitation in birds. Academic Press, San Diego, CA.

Pedersen, H.C. 1988. Territorial behaviour and breeding numbers in Norwegian Willow Ptarmigan: a removal experiment. Ornis Scandinavica 19: 81-87.

van Balen, J.H. 1973. A comparative study of the breeding ecology of the Great Tit *Parus major* in different habitats. Ardea 61: 1-93

Village, A. 1983. The role of nest site availability and territorial behaviour in limiting the breeding density of Kestrels. Journal of Animal Ecology 62: 635-645.